United States
Department of
Agriculture

Forest Service

Southern
Research Station

General Technical
Report SRS-39

Drinking Water from Forests and Grasslands

A Synthesis of the Scientific Literature

George E. Dissmeyer, Editor

The Editor:

George E. Dissmeyer, Water Quality Management—
Nonpoint Source Consultant, 145 Lake Forest Drive, NE,
Eatonton, GA 31024.

Cover photo: Hendersonville Reservoir Dam, Pisgah National Forest, North Carolina. Photo by Bill Lea.

This publication reports research involving pesticides. It does not contain recommendations for their use, nor does it imply that the uses discussed here have been registered. All uses of pesticides must be registered by appropriate State and/or Federal agencies before they can be recommended.

CAUTION: Pesticides can be injurious to humans, domestic animals, desirable plants, and fish or other wildlife—if they are not handled or applied properly. Use all herbicides selectively and carefully. Follow recommended practices for the disposal of surplus pesticides and their containers.

September 2000

USDA Forest Service
Southern Research Station
Asheville, North Carolina

Drinking Water from Forests and Grasslands

A Synthesis of the Scientific Literature

George E. Dissmeyer, Editor

"after refreshing ourselves we proceeded on to the top of the dividing ridge from which I discovered immence ranges of high mountains still to the West of us with their tops partially covered with snow. I now decended the mountain about 3/4 of a mile which I found much steeper than on the opposite side, to a handsome bold runing Creek of cold Clear water. here I first tasted the water of the great Columbia river."

—from Meriwether Lewis' journal, August 12, 1805

Table of Contents

Part II: Effects of Recreation and the Built Environment on Water Quality

Part III: Effects of Vegetation Management on Water Quality

Part IV: Effects of Grazing Animals, Birds, and Fish on Water Quality

Page

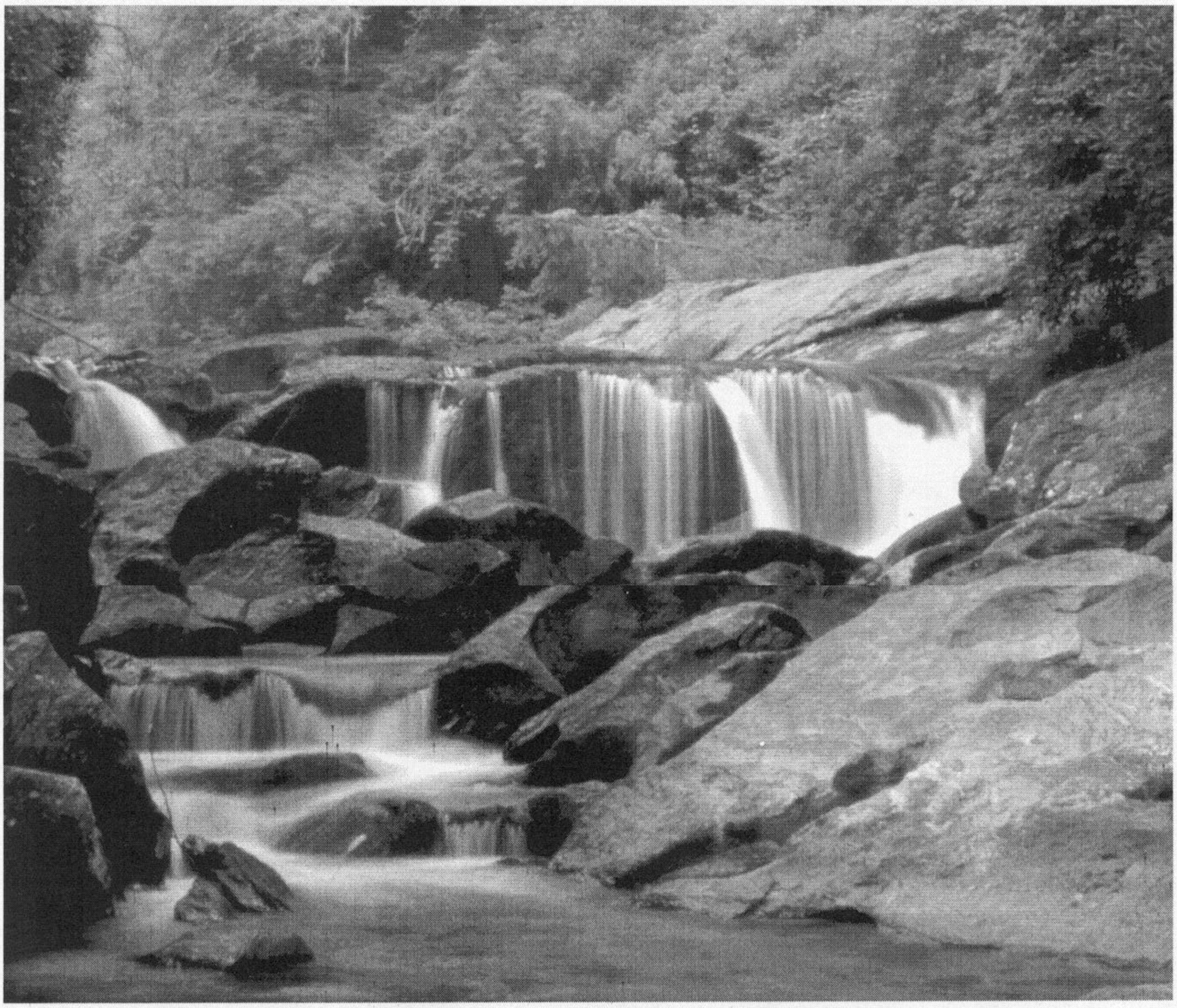
Part V: Effects of Mining and Oil and Gas Development on Water Quality

Part VI: Implications for Source Water Assessments and for Land Management and Policy

Part VII: Appendices

Part I:

Introduction



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

Executive Summary

The Safe Drinking Water Act Amendments of 1996 require every State to perform source water assessments of all public drinking water sources and make the results public by 2003. Forests and grasslands serve as sources of many public drinking water supplies, and managers of these lands are expected to participate in preparing assessments and to work with the public to assure safe drinking water. To help managers of forests and grasslands meet this requirement, this report reviews the current scientific literature about the potential of common land-use practices to introduce contaminants that pose risks to human health into public drinking water sources. Potential audiences for this report include managers of national forests and grasslands and managers of other public and private lands with similar uses. Operators of public drinking water utilities and citizens' groups concerned with drinking water may also find this report useful.

Safe drinking water is essential to protect public health. Modern drinking water treatment can reduce most contaminants in source water to acceptable levels before it is delivered to consumers, but costs increase significantly when more rigorous treatment is needed to cleanse contaminated source water. Managing land to prevent source water contamination may be more cost-effective and may better protect human health than treating water after it has been contaminated.

Water from forests and grasslands is usually cleaner than water from urban and agricultural areas. Nevertheless, many common practices on forests and grasslands can contaminate drinking water sources. Soil disturbing activities such as road construction and maintenance, forest harvesting, and intermixed urban and wildland uses can introduce sediment into drinking water sources. Disease organisms may enter source waters from: (1) recreation and other human activities that lack developed sanitary facilities, (2) malfunctioning sewage disposal facilities, and (3) wild and domestic animals concentrated near source waters. Nutrients may enter source water from fertilizer and from atmospheric deposition of nitrogen compounds. Toxic chemicals may reach source water from pest control; from extraction of minerals, oil, and gas; from accidental chemical spills along highways and utility corridors; and from leaking underground storage tanks.

Gaps exist in the scientific understanding of the effects of many land-use practices on drinking water sources. For example, pathogens in wild animal populations and their transmission to source water are poorly known. Risk of contamination from recreation that occurs in areas without developed sanitary facilities is largely unstudied. Effects of multiple land uses that overlap in time and space across large watersheds are difficult to predict with current knowledge. Managers should consider uncertainties due to these unknowns in land-use decisions until research fills these knowledge gaps.

Source water assessments for forest and grassland watersheds are not likely to be fundamentally different from those in areas with other land uses. Scientific information will need to be applied locally on a case-by-case basis to consider what natural and human activities have a reasonable potential to introduce contaminants that are likely to reach a drinking water intake. Assessments will need to integrate across conventional disciplinary boundaries to assess the overall degree of risk to drinking water sources. Scientists, land managers, and the public will need to cooperate to translate the basic information in this report into meaningful source water assessments.

Keywords: Economics, land use, nutrients, pathogens, sediments, source water assessments, toxics.

Chapter 1

Goals of this Report

Douglas F. Ryan and Stephen Glasser[1]

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

[1] Staff Watershed Specialist, Wildlife, Fish, Water, and Air Research Staff; and Water Rights and Uses Program Manager, Watershed and Air Management, USDA Forest Service, Washington, DC, respectively.

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.

Acknowledgments

We thank a number of organizations and individuals for their support in developing this document. They include the Research and Development, National Forest System, and State and Private Forestry Deputy Areas of the U.S. Department of Agriculture, Forest Service; the U.S. Environmental Protection Agency, Office of Ground Water and Drinking Water; and the National Council for Air and Stream Improvement; each of which contributed funds and expertise toward this effort. In addition, we acknowledge the contributions of the American Water Works Association, the Centers for Disease Control and Prevention, the U.S. Geological Survey, the State Forester of Massachusetts, and numerous peer reviewers.

Literature Cited

American Water Works Association. 1953. Water quality and treatment. New York: American Water Works Association, Inc. 451 p.

Dissmeyer, George E. 1994. Evaluating the effectiveness of forestry best management practices in meeting water quality goals and standards. Misc. Publ. 1520. Washington, DC: U.S. Department of Agriculture, Forest Service. 166 p.

U.S. Department of Agriculture, Forest Service. 1993. The principal laws relating to Forest Service activities. Washington, DC: U.S. Government Printing Office. 1,163 p.

Chapter 2

Drinking Water Quality

F.N. Scatena[1]

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.

[1] Ecosystem Team Leader, USDA Forest Service, International Institute of Tropical Forestry, Río Piedras, PR.

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; NH_4^{+1} = ammonium; NO_3^{-1} = nitrate; O = organic load; P = phosphorous; S = synthetic organic compounds; SO_4^{-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.
[a] An * 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
	- - - Milligrams per liter - - -			
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
	- - - Milligrams per liter - - -			
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
Heptachlorepoxide	Zero[d]	.0002	Liver damage, increased risk of cancer	Breakdown of hepatachlor
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[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* - - -			
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				
Giardia lamblia	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 *E. coli*)	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_2CO_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. Oxic 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

	Biochemical transformations[a]		Chemical reactions		Physiochemical retardation[b]	
Contaminant	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

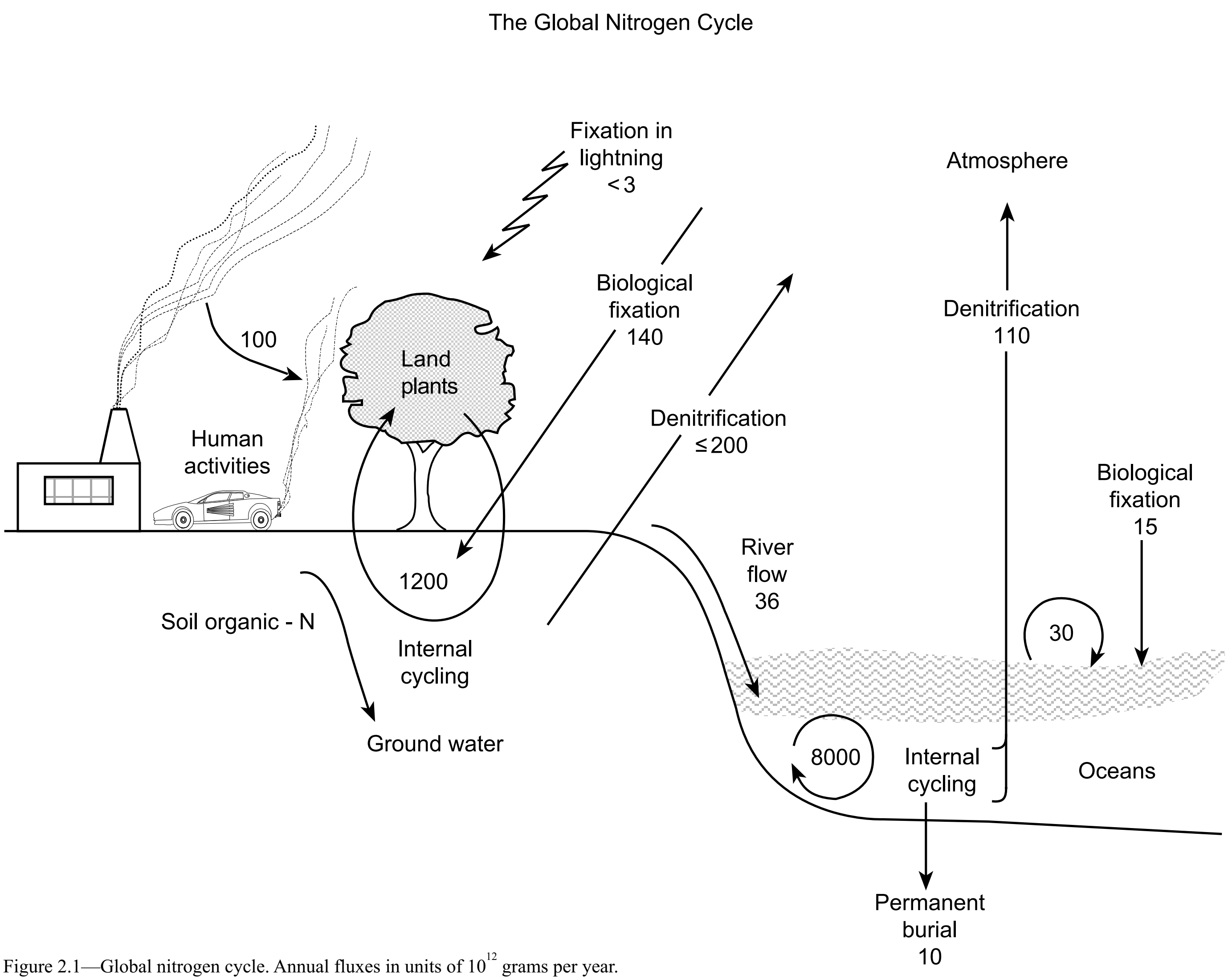


Figure 2.1—Global nitrogen cycle. Annual fluxes in units of 10^{12} grams per year.

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

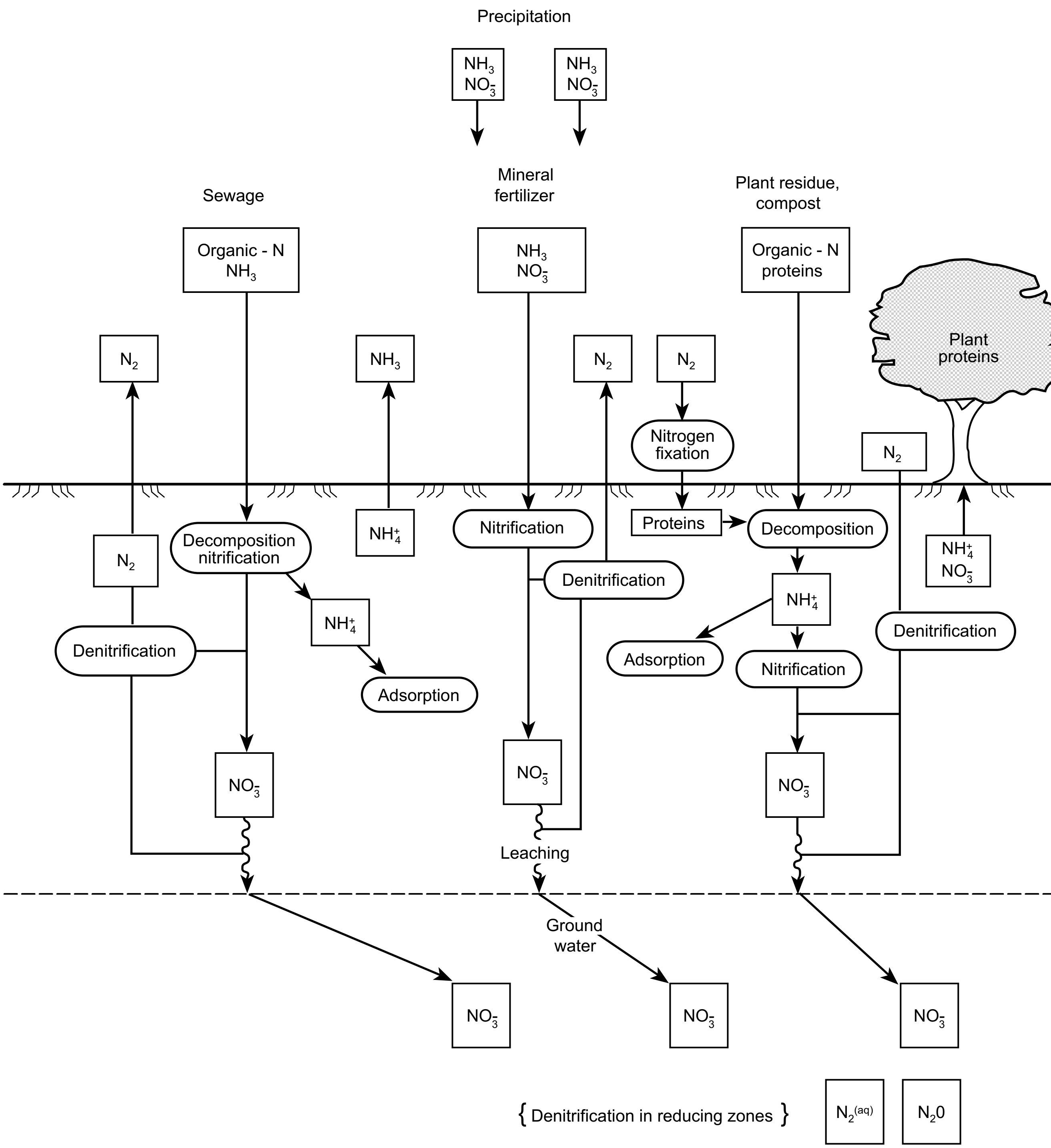


Figure 2.2—Sources and pathways of nitrogen in the subsurface environment.

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.

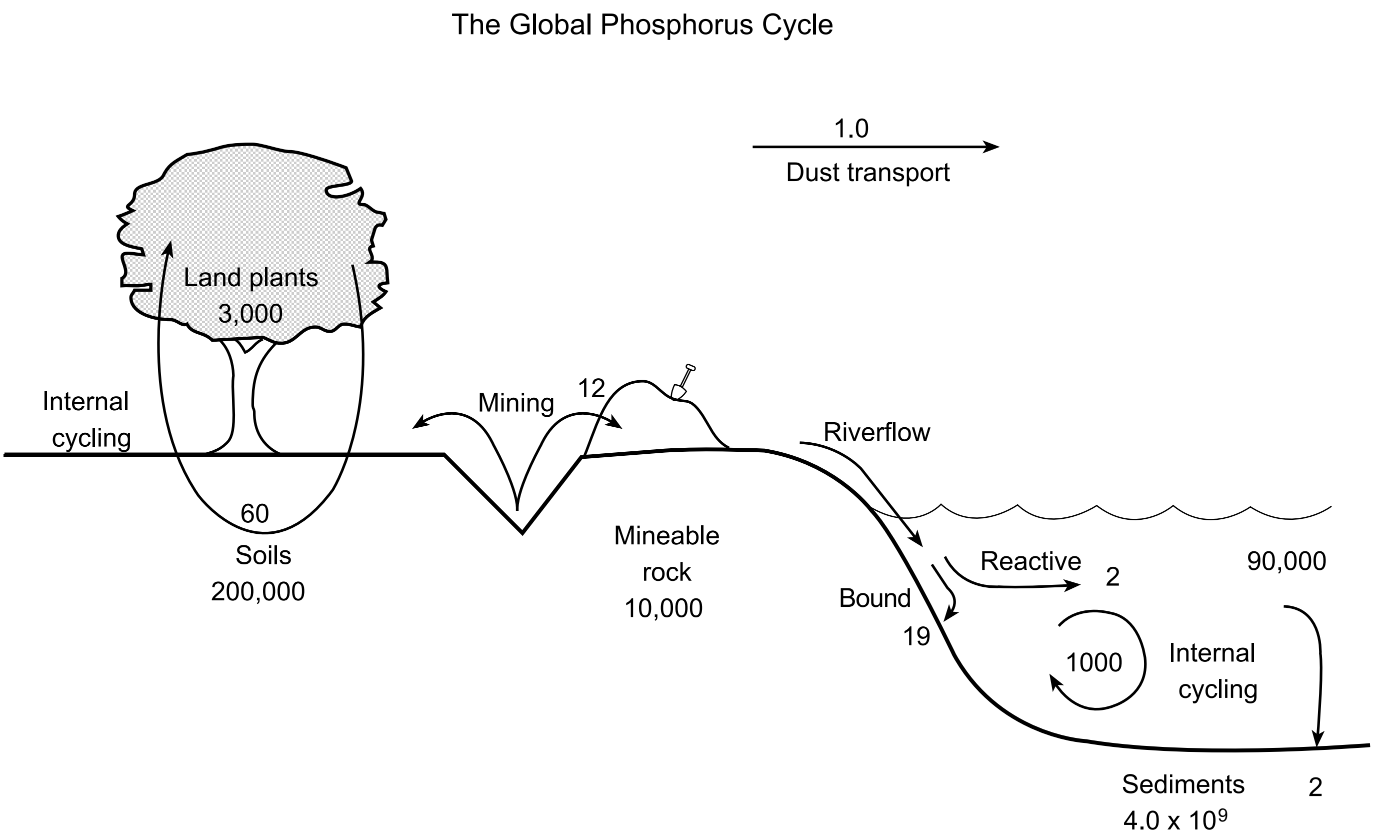


Figure 2.3—Global phosphorus cycle. Annual fluxes in units of 10^{12} grams per year.

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 *Mycobacterium avium* intracellular (MAC) *Shigella* (several strains) *Helicobacter pylori* *Vibrio cholerae* *Salmonella typhi* *S. typhimurum* Yersinia *Campylobacter* (several strains) *Escherichia coli* (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 *E. coli* (both nonpathogenic and pathogenic strains) Enterococci Fecal streptococci *Clostridium perfringens* (anaerobic spores) *Klebsiella pneumoniae* *Aeromonas hydrophila*	Bacteriophage Bacteroides phage Coliphage Male-specific coliphage FRNA phage FDNA phage Host *Salmonlla* WG-49 Host *E. coli* C-3000 Host *E. coli* FAMP Host *E. coli* 15597 Somatic coliphage Host *E. coli* C 13706, C-3000 Host *Salmonella* 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.

Literature Cited

Bannerman, R.; Owens, D.; Dodds, R.; Hornewer, N. 1993. Sources of pollutants in Wisconsin stormwater. Water Science and Technology. 28(3–5): 241–259.

Bitton, G.; Farrah, S.R.; Montague, C.L.; Akin, E.W. 1986. Viruses in drinking water. Environmental Science and Technology. 20(2): 216–222.

Boyd, R.A. 1996. Distribution of nitrate and orthophosphate in selected streams in central Nebraska. Water Resources Bulletin. 32(6): 1247–1258.

Cox, C.R. 1964. Operation and control of water treatment processes. Geneva: World Health Organization. 390 p.

Edwards, D.R.; Coyne, M.S.; Vendrell, P.F. [and others]. 1997. Fecal coliform and streptococcus concentrations in runoff from grazed pastures in northwest Arkansas. Water Resources Bulletin. 33(2): 413–422.

Fenn, M.E.; Poth, M.A.; Aber, J.D. [and others]. 1998. Nitrogen excess in North America ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecological Applications. 8(3): 706–733.

Foster, S.; Gomes, D.C. 1989. Groundwater quality monitoring: an appraisal of practices and costs. Lima, Peru: Pan American Center for Sanitary Engineering and Environmental Sciences. 103 p.

Freeze, R.A.; Cherry, J.A. 1979. Groundwater. Englewood Cliffs, NJ: Prentice-Hall, Inc. 604 p.

GEMS. 1991. Freshwater pollution. UNEP/GEMS Environment Library 6. [Place of publication unknown]: [Publisher unknown]. 36 p.

James, T.R.; Havens, K.E. 1996. Algal bloom probability in a large subtropical lake. Water Resources Bulletin. 32(5): 995–1006.

Jordan, T.E.; Corell, D.L.; Weller, D.E. 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. Journal of Environmental Quality. 22: 467–473.

Keswick, B.H.; Gerba, C.P. 1980. Viruses in ground water. Environmental Science and Technology. 14(11): 1290–1297.

Kinzig, A.P.; Socolow, R.H. 1994. Human impacts on the nitrogen cycle. Physics Today. November: 24–31.

Lewis, W.M., Jr.; Melack, J.M.; McDowell, W.H. [and others]. 1999. Nitrogen yields from undisturbed watersheds in the Americas. Biogeochemistry. 46: 149–162.

MacDonald, L.H.; Smart, A.W.; Wissmar, R.C. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. U.S. EPA 910/9–91–001. 166 p. Available from: U.S. Environmental Protection Agency, Region 10, NPS Section, WD–139, 1200 Sixth Avenue, Seattle, WA 98101.

Martin, A.B.; Cooke, G.D.; Carlson, R.E. 1993. Lake sediments as potential sources of trihalomethane precursors. Water Resources Research. 27(12): 1725–1729.

Matson, P.A.; Parton, W.J.; Power, A.G.; Swift, M.J. 1997. Agricultural intensification and ecosystem properities. Science. 77: 504–509.

Mielke, H.W. 1999. Lead in the inner cities. American Scientist. 87: 62–73.

Monzingo, D.L.; Stevens, D.R. 1986. *Giardia* contamination of surface waters: a survey of three selected backcountry streams in Rocky Mountain National Park. Water Resour. Rep. 86–2. Fort Collins, CO: National Park Service; Colorado State University. 18 p.

Nathanson, J.A. 1986. Basic environmental technology. Englewood Cliffs, NJ: Prentice-Hall, Inc. 479 p.

National Research Council. 1997. Technologies for the small system. In: Safe water from every tap. Improving water services to small communities. Water Science and Technology Board. Washington, DC: National Academy Press: 48–115.

Peterjohn, W.T.; Correll, D.L. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology. 65: 1466–1475.

Schlesinger, W.H. 1997. Biogeochemistry an analysis of global change. 2d ed. New York: Academic Press. 588 p.

Spahr, N.E.; Wynn, K.H. 1997. Nitrogen and phosphorus in surface waters of the Upper Colorado River Basin. Journal of the American Water Resources Association. 33(3): 547–560.

Stumm, W.; Morgan, J.J. 1970. Aquatic chemistry. 2d ed. New York: John Wiley. 583 p.

U.S. EPA. 1999a. Groundwater rule. [Not paged]. [Available on the Web at: http://www.epa.gov/OGWDW/].

U.S. EPA. 1999b. National primary drinking water regulations. Primary drinking water standards. [Not paged]. [Available on the Web at: http://www.epa.gov/OGWDW/wot/appa.html].

U.S. EPA. 1999c. National secondary drinking water regulations. [Not paged]. [Available on the Web at: http://www.epa.gov/OGWDW/wot/appa.html].

Vitousek, P.M.; Melillo, J.M. 1979. Nitrate losses from disturbed forests: patterns and mechanisms. Forest Science. 25: 605–619.

Vitousek, P.M.; Mooney, H.A.; Lubchenco, J.; Melillo, J.M. 1997. Human domination of the Earth's ecosystem. Science. 277: 494–499.

Ward, J.V. 1985. Thermal characteristics of running waters. Hydrobiologia. 125: 31–46.

Chapter 3

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

F.J. Swanson, F.N. Scatena, G.E. Dissmeyer, M.E. Fenn,
E.S. Verry, and J.A. Lynch[1]

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.

[1] Ecosystem Team Leader, USDA Forest Service, Pacific Northwest Research Station, Corvallis, OR; Ecosystem Team Leader, USDA Forest Service, International Institute of Tropical Forestry, Río Piedras, PR; Consultant, Water Quality Management–Nonpoint Source, Eatonton, GA; Research Plant Pathologist, USDA Forest Service, Pacific Southwest Forest Experiment Station, Forest Fire Laboratory, Riverside, CA; Research Forest Hydrologist, USDA Forest Service, North Central Experiment Station, Grand Rapids, MN; and Professor of Forest Hydrology, Pennsylvania State University, University Park, PA, respectively.

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.

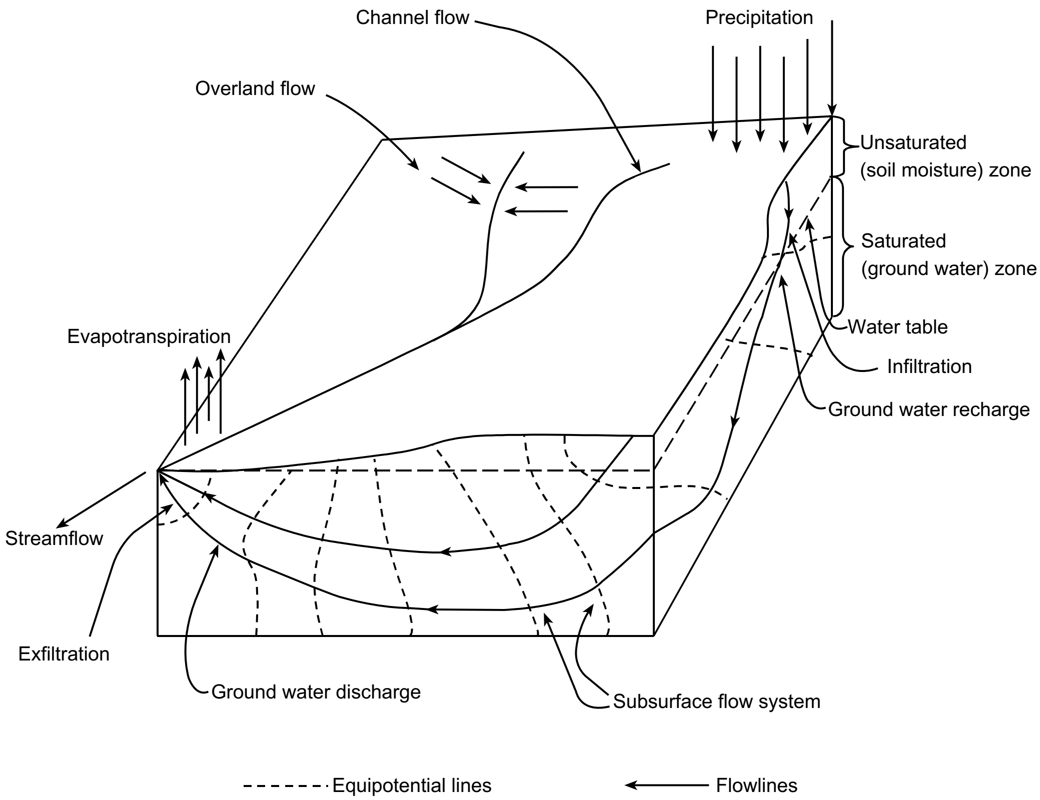


Figure 3.1—Hydrologic cycle within a watershed.

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^{-1}) 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 (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 (pH <4.0) from acid mine drainage. The sites at risk are those where precipitation pH is $\leq$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 $\leq$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).

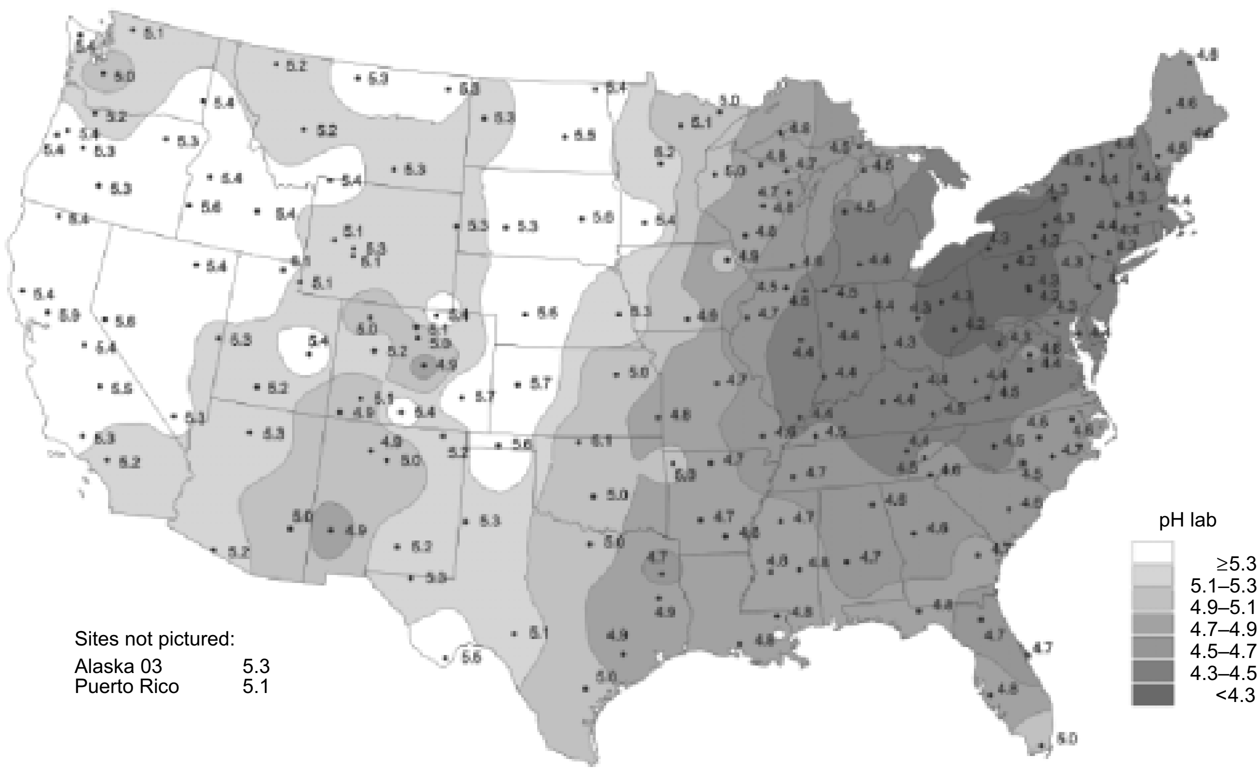


Figure 3.2—Average acidity (pH) of precipitation in the United States from 1988 through 1997 (National Atmospheric Deposition Program 1999).

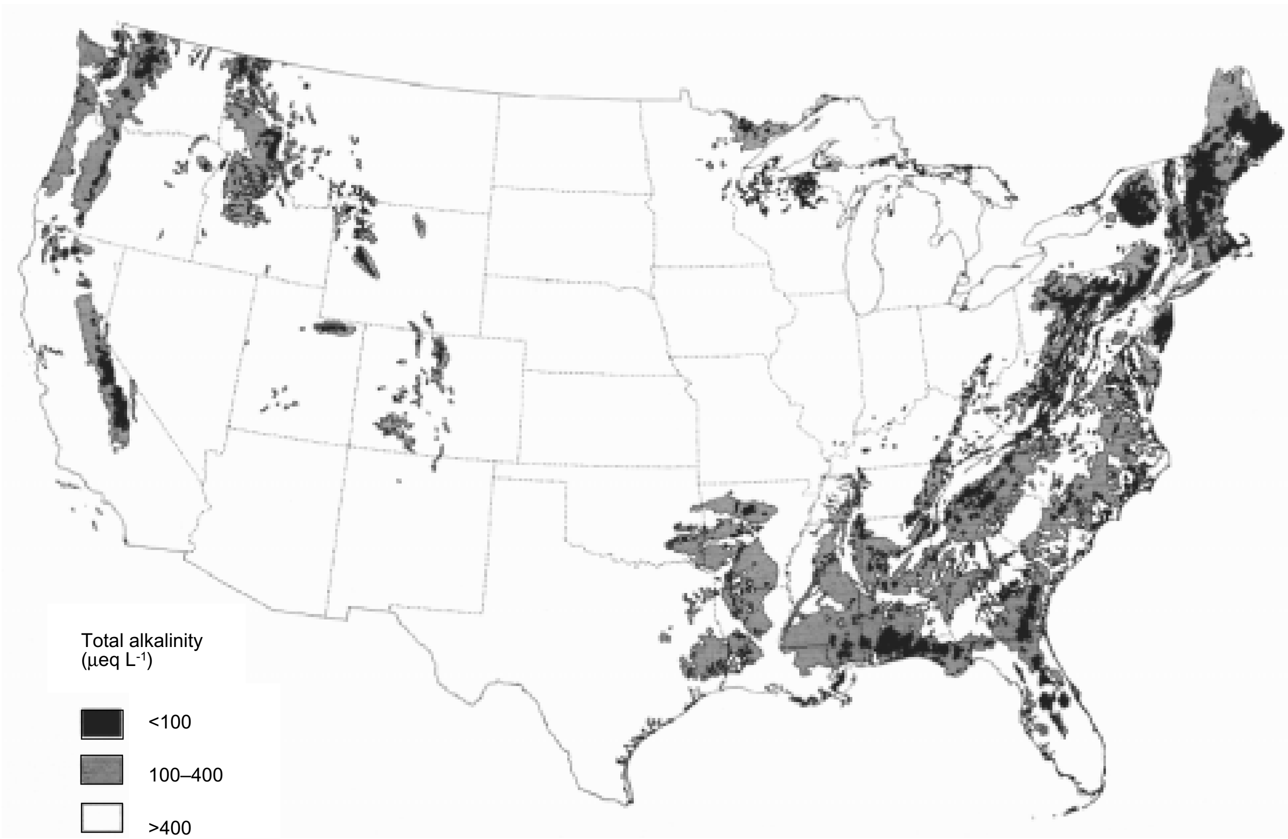


Figure 3.3—The acid buffering capacity (total alkalinity) in lake water and, by inference, the surrounding water in soils and ground water for the United States. Where lakes have low or negative alkalinities in the black areas, surface water pH values may be <4.5. (Map prepared by J.M. Omernick, G.E. 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).

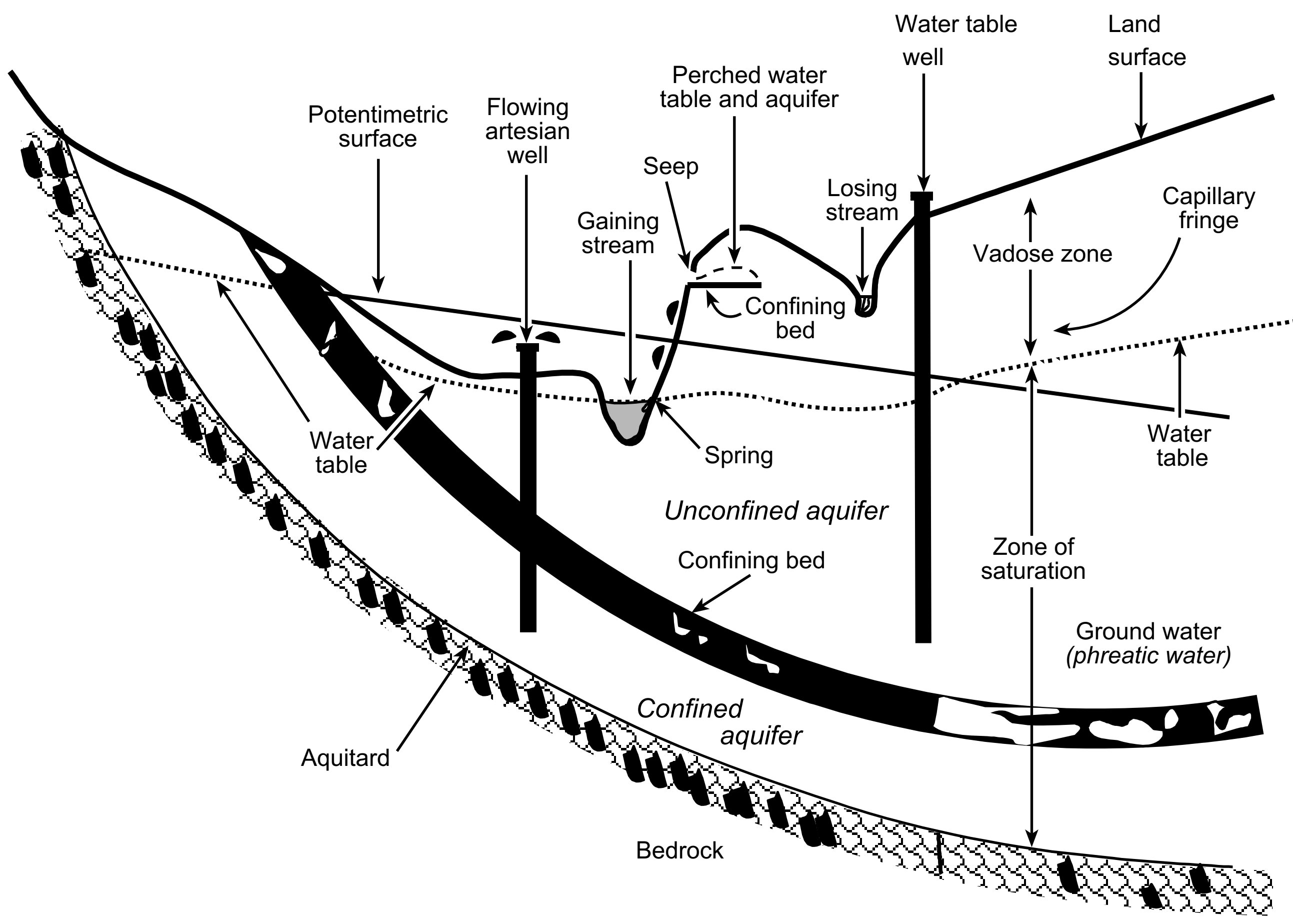


Figure 3.4—Schematic illustrating ground water terms and concepts.

Exchanges between ground water and mineral particles generally increase the concentrations of total dissolved solids (TDS), cations, calcium bicarbonate ($Ca(HCO_3)_2$), magnesium bicarbonate ($Mg(HCO_3)_2$), 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-km^2) 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.

Literature Cited

Aber, J.D.; Magill, A.; McNulty, S.G. [and others]. 1995. Forest biogeochemistry and primary production altered by nitrogen saturation. Water, Air, and Soil Pollution. 85: 1665–1670.

Aber, J.D.; Nadelhoffer, K.J.; Steudler, P.; Melillo, J.M. 1989. Nitrogen saturation in northern forest ecosystems. BioScience. 39: 378–386.

Anderson, Henry W.; Hoover, Marvin D.; Reinhart, Kenneth G. 1976. Forests and water: effects of forest management on floods, sedimentation, and water supply. Gen. Tech. Rep. PSW–18. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 115 p.

Bates, Deigh; Willis, Katherine; Swanson, Frederick [and others]. 1998. North Santiam River turbidity study, 1996–1997. Berkeley, CA: Watershed Management Council, University of California. Watershed Management Council Networker. 20 p. [Newsletter].

Berg, M.P.; Verhoef, H.A. 1998. Ecological characteristics of a nitrogen-saturated coniferous forest in the Netherlands. Biology and Fertility of Soils. 26: 258–267.

Beschta, R.L. 1990. Effects of fire on water quantity and quality. In: Walstad, J.D.; Radosevich, S.R.; Sandberg, D.V., eds. Natural and prescribed fire in Pacific Northwest forests. Corvallis, OR: Oregon State University Press: 219–232.

Binkley, Dan; Brown, Thomas C. 1993. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin. 29(5): 729–740.

Brown, W.M., III; Ritter, J.R. 1971. Sediment transport and turbidity in the Eel River Basin. U.S. Geol. Surv. Water Supply Pap. 1986. Washington, DC: U.S. Government Printing Office. 70 p.

Bunte', K.; MacDonald, L.H. 1999. Scale considerations and the detectability of sedimentary cumulative watershed effects. Tech. Bull. 776. Research Triangle Park, NC: National Council for Air and Stream Improvement. 327 p.

Burns, D.A.; Murdoch, P.S.; Lawrence, G.B.; Michel, R.L. 1998. Effect of groundwater springs on NO_3^{-1} concentrations during summer in Catskill Mountain streams. Water Resources Research. 34: 1987–1996.

Church, M.R. 1983. Analysis of trends based on historic measurements of surface water quality. In: Altshuller, A.P.; Linthurst, R.A., eds. Acidic deposition phenomenon and its effects: critical assessment review papers. EPA/6008–83/016a. Washington, DC: U.S. Environmental Protection Agency: 4–53, 4–89. Vol. 2.

Dise, N.B.; Wright, R.F. 1995. Nitrogen leaching from European forests in relation to nitrogen deposition. Forest Ecology and Management. 71: 153–161.

Dunne, Thomas; Leopold, Luna B. 1978. Water in environmental planning. New York: W.H. Freeman and Company. 818 p.

Durka, W.; Schulze, E.D.; Gebauer, G.; Voerkelius, S. 1994. Effects of forest decline on uptake and leaching of deposited nitrate determined from ^{15}N and ^{18}O measurements. Nature. 372: 765–767.

Federer, C.A.; Hornbeck, J.W.; Tritton, L.M. [and others]. 1989. Long-term depletion of calcium and other nutrients in Eastern U.S. forests. Environmental Management. 13: 593–601.

Fenn, M.E.; Poth, M.A. 1998. Indicators of nitrogen status in California forests. In: Bytnerowicz, A.; Arbaugh, M.J.; Schilling, S.L., tech. coords. Proceedings of the international symposium on air pollution and climate change effects on forest ecosystems; 1996 February 5–9; Riverside, CA. Gen. Tech. Rep. PSW–GTR–166. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 123–130.

Fenn, M.E.; Poth, M.A. 1999. Temporal and spatial trends in streamwater nitrate concentrations in the San Bernardino Mountains, southern California. Journal of Environmental Quality. 28: 822–836.

Fenn, M.E.; Poth, M.A.; Aber, J.D. [and others]. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecological Applications. 8: 706–733.

Fleischer, S.; Stibe, L.; Leonardson, L. 1991. Restoration of wetlands as a means of reducing nitrogen transport to coastal waters. Ambio. 20: 271–272.

Foster, D.R.; Aber, J.D.; Melillo, J.M. [and others]. 1997. Forest response to disturbance and anthropogenic stress. BioScience. 47(7): 437–445.

Foster, N.W.; Hazlett, P.W.; Nicolson, J.A.; Morrison, I.K. 1989. Ion leaching from a sugar maple forest in response to acidic deposition and nitrification. Water, Air, and Soil Pollution. 48: 251–261.

Fredriksen, R.L.; Moore, D.G.; Norris, L.A. 1975. The impact of timber harvest, fertilization, and herbicide treatment on streamwater quality in western Oregon and Washington. In: Bernier, B.; Winget, C.H., eds. Forest soils and forest land management: Proceedings of the 4th North American forest soils conference; 1973 August; Quebec, Canada. Quebec: Les Presses de IÕUniversite Laval: 283–313.

Freeze, R.A.; Cherry, J.A. 1979. Groundwater. Englewood Cliffs, NJ: Prentice-Hall, Inc. 604 p.

Gray, Lawrence J.; Macpherson, Gwendolyn L.; Koelliker, James K.; Dodds, Walter K. 1998. Hydrology and aquatic chemistry. In: Knapp, Alan K.; Briggs, John M.; Hartnett, David C.; Collins, Scott L., eds. Grassland dynamics: long-term ecological research in tallgrass prairie. New York: Oxford University Press, Inc. 364 p.

Gupta, R.S. 1995. Hydrology and hydraulic systems. Prospect Heights, IL: Waveland Press. 739 p.

Hack, J.T.; Goodlett, J.C. 1960. Geomorphology and forest ecology of a mountain region in the central Appalachians. U.S. Geol. Surv. Prof. Pap. 347. Reston, VA: U.S. Geological Survey. 66 p.

Haycock, N.E.; Pinay, G.; Walker, C. 1993. Nitrogen retention in river corridors: European perspective. Ambio. 22: 340–346.

Hendrickson, O.Q.; Chatarpaul, L.; Burgess, D. 1989. Nutrient cycling following whole-tree and conventional harvest in northern mixed forest. Canadian Journal of Forest Research. 19: 725–735.

Hunsaker, C.T.; Levine, D.A.; Timmins, S.P. [and others]. 1992. Landscape characterization for assessing regional water quality. In: McKenzie, D.H.; Hyatt, D.E.; McDonald, V.J., eds. Ecological indicators. London; New York: Elsevier Applied Science: 997–1006.

Johnson, K. Norman; Swanson, Frederick; Herring, Margaret; Greene, Sarah, eds. 1999. Bioregional assessments: science at the crossroads of management and policy. Washington, DC: Island Press. 395 p.

Kelsey, H.M. 1980. A sediment budget and an analysis of geomorphic process in the Van Duzen River Basin, north coastal California, 1941–1975. Geological Society of America Bulletin, Part II. 91: 1119–1216.

Likens, Gene E.; Bormann, F. Herbert. 1995. Biogeochemistry of a forested ecosystem. New York: Springer-Verlag. 160 p.

Linsley, R.K.; Kohler, M.A.; Paulhus, J.L.H. 1982. Hydrology for engineers. Hydrology and water quality. New York: McGraw-Hill: 412–442. Chapter 15.

Madej, M.A.; Ozaki, V. 1996. Channel response to sediment wave propagation and movement, Redwood Creek, CA. Earth Surface Processes and Landforms. 21: 911–927.

Marzolf, E.R.; Mulhollan, P.J.; Steinman, A.D. 1994. Improvements to the diurnal upstream-downstream dissolved oxygen technique for determining whole-stream metabolism in small streams. Canadian Journal of Fisheries Aquatic Science. 51: 1551–1599.

McDonald, M.E. 1985. Acid deposition and drinking water. Environmental Science and Technology. 19: 772–776.

McNulty, S.G.; Aber, J.D.; Newman, S.D. 1996. Nitrogen saturation in a high elevation New England spruce-fir stand. Forest Ecology and Management. 84: 109–121.

Megahan, W.F.; Platts, W.S.; Kulesza, B. l980. Riverbed improves over time: South Fork Salmon. In: Proceedings of a symposium on watershed management; 1980 July 21–23; Boise, ID. New York: American Society of Civil Engineers: 380–395. Vol. 1.

Murdoch, P.S.; Stoddard, J.L. 1992. The role of nitrate in the acidification of streams in the Catskill Mountains of New York. Water Resources Research. 28: 2707–2720.

National Atmospheric Deposition Program. 1999. National atmospheric deposition program (NRSP–3)/national trends network. (1997). Champaign, IL: Illinois State Water Survey. Map series 1999. [Map scale, size, and whether color or b/w unknown].

Newbold, J.D.; O'Neill, R.V.; Elwood, J.W.; Van Winkle, W. 1982. Nutrient spralling in streams: implications for nutrient limitations and invertebrate activity. American Naturalist. 120: 628–652.

Nriagu, J.O.; Pacyna, J.M. 1988. Quantitative assessment of worldwide contamination of air, water, and soils with trace metals. Nature. 333: 134–139.

Park, C.C. 1987. Acid rain: rhetoric and reality. London: Methuen and Co. 272 p.

Phillips, J.D. 1993. Pre- and post-colonial sediment sources and storage in the Lower Neuse Basin, North Carolina. Physical Geography. 14(3): 272–284.

Poff, N.L.; Allan, J.D.; Bain, M.B. [and others]. 1997. The natural flow regime. BioScience. 47: 769–784.

Regional Ecosystem Office. 1995. Ecosystem analysis at the watershed scale. Version 2.2. Portland, OR: Regional Ecosystem Office. 1995–689–120/21215 Region 10. Washington, DC: U.S. Government Printing Office. 26 p.

Reid, L.M.; Dunne, T. 1996. Rapid construction of sediment budgets for drainage basins. Cremlingen, Germany: Catena-Verlag. 160 p.

Reid, Leslie M. 1993. Research and cumulative watershed effects. Gen. Tech. Rep. PSW–GTR–141. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 118 p.

Reid, Leslie M. 1998. Cumulative watershed effects: Caspar Creek and beyond. In: Ziemer, Robert R., tech. coord. Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW–GTR–168. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 149 p.

Riggan, P.J.; Lockwood, R.N.; Jacks, P.M. [and others]. 1994. Effects of fire severity on nitrate mobilization in watersheds subject to chronic atmospheric deposition. Environmental Science and Technology. 28: 369–375.

Schlesinger, W.H. 1997. Biogeochemistry: an analysis of global change. 2d ed. New York: Academic Press. 588 p.

Sidle, R.C.; Pearce, A.J.; O'Loughlin, C.L. 1985. Soil mass movement—influence of natural factors and land use. Water Resour. Monog. 11. Washington, DC: American Geophysical Union. 140 p.

Stevens, P.A.; Harrison, A.F.; Jones, H.E. [and others]. 1993. Nitrate leaching from a Sitka spruce plantation and the effect of fertilization with phosphorus and potassium. Forest Ecology and Management. 58: 233–247.

Stoddard, J.L. 1994. Long-term changes in watershed retention of nitrogen: its causes and aquatic consequences. In: Baker, L.A., ed. Environmental chemistry of lakes and reservoirs. Advances in Chem. Ser. 237. Washington, DC: American Chemical Society: 223–284.

Swank, W.T.; Crossley, C.A., Jr., eds. 1988. Forest hydrology and ecology at Coweeta. New York: Springer-Verlag. 469 p.

Swanson, Frederick J. 1981. Fire and geomorphic processes. In: Mooney, H.A., ed. Proceedings of the conference on fire regimes and ecosystems; 1979 December 11–15; Honolulu. Gen. Tech. Rep. WO–26. Washington, DC: U.S. Department of Agriculture, Forest Service: 401–420.

Swanson, Frederick J.; Benda, Lee E.; Duncan, Stanley H. [and others]. 1987. Mass failures and other processes of sediment production in Pacific Northwest forest landscapes. In: Salo, Ernest O.; Cundy, Terrance W., eds. Streamside management: forestry and fishery interactions: Proceedings of the symposium; 1986 February 12–14; Seattle. Seattle: University of Washington, Institute of Forest Resources: 9–38.

Swanson, Frederick J.; Janda, R.J.; Dunne, T.; Swanston, D.N., eds. 1982. Sediment budgets and routing in forested drainage basins. Gen. Tech. Rep. PNW–141. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 165 p.

Taskey, R.D.; Harward, M.E.; Youngberg, C.T. 1978. Relationship of clay mineralogy to landscape stability. In: Youngberg, Chester T., ed. Forest soils and land use: Proceedings of the 5th North American forest soils conference; 1978 August; Fort Collins, CO. Fort Collins, CO: Colorado State University, Department of Forest and Wood Sciences: 140–164.

Trimble, S.W. 1969. Culturally accelerated sedimentation on the middle Georgia Piedmont. Athens, GA: University of Georgia. 110 p. M.S. thesis.

Trimble, S.W. 1974. Man-induced soil erosion on the southern Piedmont 1700–1970. Akeny, IA: Soil Conservation Society of America. 180 p.

Trimble, S.W. 1999. Decreased rates of alluvial sediment storage in the Coon Creek Basin, Wisconsin, 1975 to 1999. Science. 285: 1244–1246.

U.S. General Accounting Office. 1998. Oregon watersheds: many activities contribute to increased turbidity during large storms. GAO/RCED–98–220. Washington, DC: U.S. General Accounting Office. 69 p.

Vannote, R.L.; Minshall, G.W.; Cummins, K.W. [and others]. 1980. The river continum concept. Canadian Journal of Fisheries and Aquatic Science. 37: 130–137.

Viessman, W.; Knapp, J.W.; Lewis, G.L.; Harbaugh, T.E. 1977. Introduction to hydrology. New York: Harper and Row. 704 p.

Washington Forest Practices Board. 1995. Standard methodology for conducting watershed analysis under chapter 222–2 WAC. Version 3.0. Olympia, WA: Forest Practices Division, Department of Natural Resources. 673 p.

Wenger, K.F., ed. 1994. Forestry handbook. 2d ed. Washington, DC: Society of American Foresters; John Wiley. 1,334 p.

Williams, M.W.; Baron, J.S.; Caine, N. [and others]. 1996. Nitrogen saturation in the Rocky Mountains. Environmental Science and Technology. 30: 640–646.

Winter, T.C. 1981. Uncertainties in estimating the water balance of lakes. Water Resources Bulletin. 17(1): 82–115.

Winter, Thomas C.; Harvey, Judson W.; Franke, O. Lehn; Alley, William M. 1998. Ground water and surface water: a single resource. U.S. Geol. Surv. Circ. 1139. Denver: U.S. Geological Survey. 79 p.

Wolman, M.G.; Miller, J.P. 1960. Magnitude and frequency of forces in geomorphic processes. Journal of Geology. 68(1): 54–74.

World Bank. 1993. Water resources management. A World Bank policy paper. Washington, DC: The World Bank. 140 p.

Youngberg, C.T.; Harward, M.E.; Simonson, G.H.; Rai, D. 1975. Nature and causes of stream turbidity in a mountain watershed. In: Bernier, B.; Winget, C.H., eds. Forest soils and forest land management: Proceedings of the 4th North American forest soils conference; 1973 August; Quebec, Canada. Quebec: Les Presses de lÕUniversite Laval: 267–282.

Ziemer, R.R.; Lewis, J.; Lisle, T.E.; Rice, R.M. 1991. Long-term sedimentation effects of different patterns of timber harvesting: Proceedings of a symposium held during the 20th general assembly of the international union of geodesy and geophysics in Vienna, Austria; 1991 August 11–24; Vienna, Austria. Wallingford, Oxfordshire, UK: International Association of Hydrological Sciences Press: 143–150.

Chapter 4

Economic Issues for Watersheds Supplying Drinking Water

Thomas C. Brown[1]

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.

---

[1] Economist, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

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.

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 (Xd) to a level at or below the standard (Xs):

$$Xd_j \leq Xs \tag{1}$$

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

$$Xd_j = Xr_j - T_j \tag{2}$$

[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).

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.

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):

$$Xr_j = \sum_i e_i \alpha_{ij} / Q_i \tag{3}$$

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.

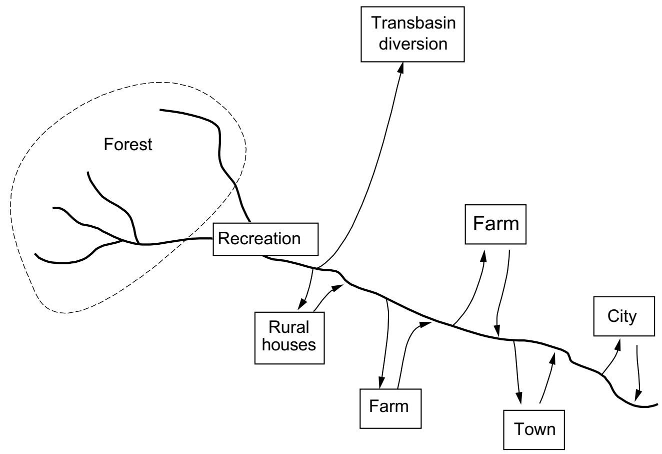


Figure 4.1—Hypothetical river basin.

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:

$$C_j = \sum_i Ce_{ij} + \sum_k Cu_{kj} + Ct_j \tag{4}$$

[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_j. 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.

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

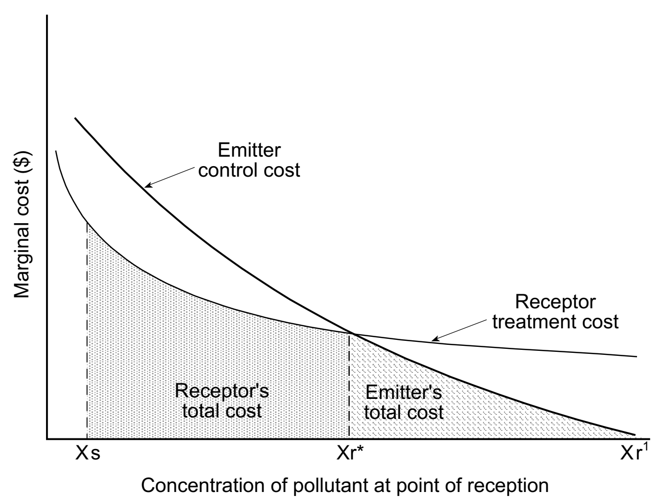


Figure 4.2—Efficient allocation of cost of meeting drinking water quality standard, with one emitter and one receptor.

[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 irritation 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.

materials necessary to meet a water-quality threat at all times. Even if the water-quality standard was set as low as Xr', the provider would still need to maintain the daily capability of handling water withdrawals with pollutant concentrations that exceeded Xr'. 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 Xs, the efficient allocation of treatment and control costs is indicated by the intersection of the two cost curves at a concentration of Xr*. To the right of Xr*, 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 Xr* costs the emitter more than it saves the receptor, and not requiring the emitter to reduce the concentration to at least Xr* 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 Xr*, and the receptor's total cost is the area below the receptor's marginal cost curve to the left of Xr* and right of Xs.

Although the receptor's marginal cost curve will always be above the emitter's at concentration level Xr' (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 Gumerman 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 Xr) or tightening of water-quality standards (lowering of Xs). 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 (Xs) of the drinking water provider, as in figure 4.4. As

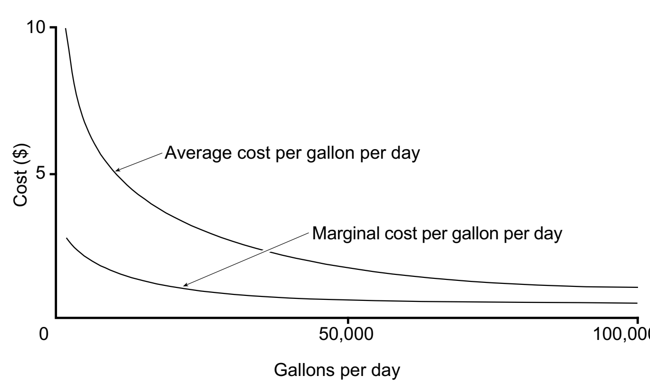


Figure 4.3—Construction cost of pressure filtration plant with an infiltration rate of 2 gallons per minute per square foot (Gummerman and others 1978).

[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:

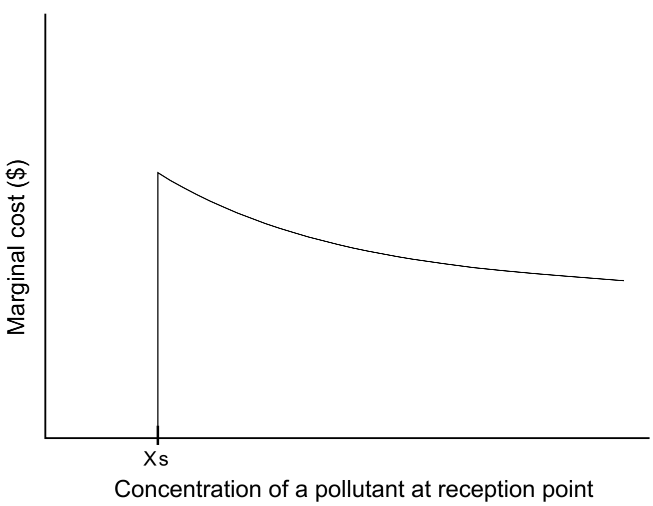


Figure 4.4—Long-term marginal cost as a function of pollutant concentration.

- 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 Xr. 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.
- Xr 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.

[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.

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 (km^2)], 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).

Literature Cited

Adler, Robert W. 1992. Returning to the goals of the Clean Water Act. Water Resources Update. 88: 2–30.

Anderson, Robert C.; Carlin, Alan; McGartland, Albert M.; Weinberger, Jennifer B. 1997. Cost savings from the use of market incentives for pollution control. In: Kosobud, Richard F.; Zimmerman, Jennifer M.,eds. Market-based approaches to environmental policy: regulatory innovations to the fore. New York: Van Nostrand Reinhold: 15–46.

Ashendorff, Arthur; Principe, Michael A.; Seeley, Anne [and others]. 1997. Watershed protection for New York City's supply. Journal of the American Water Works Association. 89(3): 75–88.

Baumol, William J.; Oates, Wallace E. 1975. The theory of environmental policy. Englewood Cliffs, NJ: Prentice-Hall, Inc. 272 p.

Binkley, Dan; Brown, Thomas C. 1993a. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin. 29(5): 729–740.

Binkley, Dan; Brown, Thomas C. 1993b. Management impacts on water quality of forests and rangelands. Gen. Tech. Rep. RM–239. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 115 p.

Braden, John B.; Johnson, Gary V.; Bouzaher, Aziz; Miltz, David. 1989. Optimal spatial management of agricultural pollution. American Journal of Agricultural Economics. 71(2): 404–413.

Brown, Thomas C.; Brown, Douglas; Binkely, Dan. 1993. Laws and programs for controlling nonpoint source pollution in forest areas. Water Resources Bulletin. 29(1): 1–13.

Chang, Ching-Cheng; Atwood, Jay D.; Alt, Klaus; McCarl, Bruce A. 1994. Ecomonic impacts of erosion management measures in coastal drainage basins. Journal of Soil and Water Conservation. 49(6): 606–611.

Easter, K. William. 1993. Differences in the transaction costs of strategies to control agricultural offsite and undersite damages. In: Russell, Clifford S.; Shogren, Jason F., eds. Theory, modeling, and experiences in the management of nonpoint-source pollution. Boston: Kluwer Academic Publishers: 37–69.

Freeman, A. Myrick, III. 1982. Air and water pollution control: a benefit-cost assessment. New York: John Wiley. 186 p.

Freeman, A. Myrick, III. 1990. Water pollution policy. In: Portney, Paul R.; Dower, Roger C.; Freeman, A. Myrick [and others], eds. Public policies for environmental protection. Washington, DC: Resources for the Future: 87–149.

Freeman, A. Myrick, III. 1993. The measurement of environmental and resource values: theory and methods. Washington, DC: Resources for the Future. 516 p.

Freeman, A. Myrick, III; Haveman, Robert H.; Kneese, Allen V. 1973. The economics of environmental policy. New York: John Wiley. 184 p.

Gianessi, Leonard; Peskin, Henry M. 1981. Analysis of national water pollution control policies: 2. agricultural sediment control. Water Resources Research. 17(4): 804–821.

Gumerman, Robert C.; Culp, Russell L.; Hansen, Sigurd. 1978. Estimating costs for water treatment as a function of size and treatment plant efficiency. EPA–600/2–78–182. Cincinnati: U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory. 282 p.

Hertzler, Patricia C.; Davies, Clive. 1997. The cost of infrastructure needs. Journal of the American Water Works Association. 89(3): 55–61.

Johnson, Edwin L. 1967. A study in the economics of water quality management. Water Resources Research. 3(2): 291–305.

Kneese, Allen V.; Bower, Blair T. 1968. Managing water quality: economics, technology, and institutions. Washington, DC: Resources for the Future. 328 p.

Kneese, Allen V.; Bower, Blair T. 1979. Environmental quality and residuals management. Baltimore: Johns Hopkins University Press. 337 p.

Kozloff, Keith; Taff, Steven J.; Wang, Yingmin. 1992. Microtargeting the acquisition of cropping rights to reduce nonpoint source water pollution. Water Resources Research. 28(3): 623–628.

Lettenmaier, Dennis; Hooper, Eric R.; Wagoner, Colin; Faris, Kathleen B. 1991. Trends in stream quality in the continental United States, 1978–1987. Water Resources Research. 27(3): 327–339.

Lyon, Randolph; Farrow, Scott. 1995. An economic analysis of Clean Water Act issues. Water Resources Research. 31(1): 213–223.

Malik, Arun S.; Larson, Bruce A.; Ribaudo, Marc. 1994. Economic incentives for agricultural nonpoint source pollution control. Water Resources Bulletin. 30(3): 471–479.

Moore, Walter B.; McCarl, Bruce A. 1987. Off-site costs of soil erosion: a case study in the Williamette Valley. Western Journal of Agricultural Economics. 12(1): 42–49.

Okun, Daniel A.; Craun, Gunther F.; Edzwald, James K. [and others]. 1997. New York City: to filter or not to filter. Journal of the American Water Works Association. 89(3): 62–74.

Raucher, Robert S.; Castillo, Eloise Trabka; Dixon, Ann [and others]. 1995. Estimating the cost of compliance with drinking water standards: a user's guide. Denver: American Water Works Association Research Foundation. 380 p.

Ribaudo, Marc O. 1989. Water quality benefits from the conservation reserve program. Agric. Econ. Rep. 606. Washington, DC: U.S. Department of Agriculture, Economic Research Service. 30 p.

Schleich, Joachim; White, David; Stephenson, Kurt. 1996. Cost implications in achieving alternative water quality targets. Water Resources Research. 32(9): 2879–2884.

Segerson, Kathleen. 1988. Uncertainty and incentives for nonpoint pollution control. Journal of Environmental Economics and Management. 15: 87–98.

Segerson, Kathleen. 1990. Incentive policies for control of agricultural water pollution. In: Braden, John B.; Lovejoy, Stephen B., eds. Agriculture and water quality: international perspectives. Boulder, CO: Lynne Rienner Publishers: 39–61.

Shortle, James S. 1987. Allocative implications of comparisons between the marginal costs of point and nonpoint source pollution abatement. Northeastern Journal of Agricultural and Resource Economics. 16: 17–23.

Shortle, James; Abler, David G. 1997. Nonpoint pollution. In: Folmer, Henk; Tietenberg, Tom, eds. The international yearbook of environmental and resource economics 1997/1998: a survey of current issues. Cheltenham, UK: Edward Elgar Publishing Limited: 114–155.

Shortle, James S.; Dunn, James W. 1986. The relative efficiency of agricultural source water pollution control policies. American Journal of Agricultural Economics. 68(3): 668–677.

Singh, Jasbinder; Stein, Raffael; Chandra, Sanjay; Snyder, Brett. 1988. The municipal sector study. EPA 230–09/88–038. Washington, DC: U.S. Environmental Protection Agency. 166 p.

Tietenberg, T.H. 1985. Emissions trading: an exercise in reforming pollution policy. Washington, DC: Resources for the Future. 222 p.

Tietenberg, Tom. 1988. Environmental and natural resource economics. Glenview, IL: Scott, Foresman, and Company. 559 p.

U.S. EPA. 1987. National water quality inventory: 1986 report to Congress. EPA–440/4–87–008. Washington, DC: U.S. Environmental Protection Agency. 141 p.

U.S. EPA. 1998. National water quality inventory: 1996 report to Congress. EPA–841–R–97–008. Washington, DC: U.S. Environmental Protection Agency. 521 p.

Young, Douglas L.; Walker, David J.; Kanjo, Paul L. 1991. Cost effectiveness and equity aspects of soil conservation programs in a highly erodible region. American Journal of Agricultural Economics. 73(4): 1053–1062.

Part II:

Effects of Recreation and the Built Environment on Water Quality

Canoeists at Ozark Landing, Buffalo National Wild and Scenic River, Buffalo National River, Arkansas. Photo by Bill Lea

Chapter 5

Hydromodifications—Dams, Diversions, Return Flows, and Other Alterations of Natural Water Flows

Stephen P. Glasser[1]

Introduction

The term hydromodification is commonly used to describe all activities, which alter the natural flow of water. This chapter addresses the effects of structures, such as dams, headgates, reservoirs, canals, water wells, diversion ditches, and flumes upon the quality of raw drinking water before it arrives at the water treatment plant. It also includes a discussion of the effects of land application of treated sewage sludge, return flows, wetland modifications, and reclaiming wastewater upon drinking water quality.

Issues and Risks

The U.S. Environmental Protection Agency (EPA) ranked hydromodification as the third leading cause of water-quality impairment to rivers. Only agriculture and municipal sewage treatment plants ranked higher (U.S. EPA 1995). Nationwide, there are over 68,000 medium and large dams built for hydropower, water supply, flood control, and other purposes. The U.S. Geological Survey estimates the cumulative storage capacity of these dams is almost 450 million acre-feet [550 billion meters (m^3)].[2] The Bureau of Reclamation manages about 600 dams and 53,000 miles [85 000 kilometers (km)] of canals in 17 Western States; the Army Corps of Engineers has about 700 dams and accounts for about one-third of all water in storage in the Nation (Reetz and others 1998). There are about 2,350 dams with a total storage capacity of about 55 million acre-feet (68 billion m^3) on land administered by the U.S. Department of Agriculture, Forest Service. Half are owned and operated by the Agency mostly for recreation, fire protection, and fish or wildlife needs. The others are owned and operated by other Federal agencies, States, and private parties under special-use authorizations, mostly for irrigation, recreation, and water supply.[3]

There are also thousands of small dams in the United States that were designed and built to store drinking water during periods when inflows to the reservoir are greater than the water removed from the reservoir. Some of these reservoirs were built and are still operated solely to provide a reliable water supply. Since the 1940's, however, some of the existing ones and almost all new reservoirs became multi-purpose; that is, they serve recreation, irrigation, flood control, and sometimes hydropower needs, while supplying drinking water. Often these other purposes create water-quality problems for human health by altering water temperature, sediment transport, biological oxygen demand, chemical oxygen demand, total dissolved solids, and streamflow. Related information on these problems can be found in chapters 2 and 3.

The diversion and transport of water from one watershed to another can result in physical, chemical, and microbiological contamination of the receiving waterbody and cause channel erosion, sediment transport, and deposition in reservoirs and channels. Subsequent dredging in large rivers and reservoirs often accelerates downcutting of headwater streams and destabilizes streambanks, even where stream gradients are quite flat, such as in Mississippi.

Drainage of wetlands with ditches is a form of hydromodification that can change water chemistry by adding organic compounds, thereby affecting water treatment processes and costs. Application of treated sewage sludge to forested land has been evaluated for its risk of contaminating water supplies with pathogens and found to be a low risk in most situations. Reclaiming sewage effluent water for drinking water is done in other countries, but is not yet commonplace in the United States.

Nearly all these hydromodifications are influenced by water rights laws which vary considerably from State to State. In most Western States, laws require water users to divert water out of streams or rivers to obtain a State water right. This removal often results in higher water temperatures, lower oxygen levels, reduced sediment transport capacity, and other water-quality problems in the remaining water

[1] Water Rights and Uses Program Manager, USDA Forest Service, Washington, DC.
[2] Personal communication. 1999. Robert Hirsch, Chief Hydrologist, U.S. Geological Survey, 807 National Center, Reston, VA 20192.
[3] Personal communication. 1998. James Padgett, Chief Hydraulic Engineer, USDA Forest Service, Washington, DC 20250.

(Getches and others 1991). The riparian water rights doctrine used in most States east of the 100th meridian generates water-quality problems because most of the water is returned to the channel.

Findings

Hydromodifications can impact water quality via algae blooms, trihalomethane production, sediment transport and deposition, and changes in chemical, physical, and microbiological properties.

Effects of Dams and Impoundments on Water Quality

The size and depth of impoundments and the residence time of water within them can affect water quality chemically, physically, and biologically. As water flows into a reservoir, its velocity slows, reducing the diffusion of oxygen from the air into the surface water. In turn, biological and chemical oxygen demands may deplete oxygen, especially near the bottom. This phenomenon has been well studied, and detailed models that quantify this effect have been developed. Anoxic conditions generally cause secondary problems in drinking water, usually taste, smell, color, and increased concentrations of iron, manganese, and sulfide. These problems usually do not pose health risks, but may increase water treatment costs. Under some conditions, impoundments can cause toxic algae blooms which can pose health risks.

Case Study: Toxic Algae Bloom at Hebgen Lake, MT

The operation of dams can affect the likelihood of blue-green algae blooms, which sometimes produce toxins that have been reported to be fatal to livestock, wildlife, and pets, and pose risk to human health. For example, Hebgen Lake, Gallatin National Forest, MT, experienced a toxic algae bloom in June 1977 (Juday and others 1981). A family camping at the Forest Service campground on the Grayling Arm of Hebgen Lake (actually a medium-size reservoir) was hysterical after their pet dog went into convulsions after drinking some of the lake water. Their dog died a few minutes later. When Forest Service personnel and a Gallatin County sanitarian arrived at the campground, they were besiged by people frightened by what they had witnessed that day. Several more pet dogs had died, and everyone could see the bodies of dead cattle lying near the lakeshore beyond the campgound fence. A green scum was on the surface of the water that was different from the algae seen in previous years. This coating resembled thick, green pea soup, was odorless, and went at least 50 feet (15 m) offshore. Water samples were taken, including the green algae, and packed in ice. The sanitarian posted his Area Closed signs at the campground and it was closed down that day.

The next day the samples were taken to the State Water Quality Bureau scientists in Helena. After they heard what had been found, they agreed to go to Hebgen Lake with Forest Service personnel. They phoned some toxic algae experts and reported this episode. These experts arrived a few days later and began intensive studies of the algal bloom. They identified the culprit as *Anabaena flos-aquae*, a blue-green alga that sometimes produces a very potent toxin (anatoxin-a), which is released into the water. No human deaths have been attributed to anatoxin-a poisoning, but over the past 100 years, the number of domestic and wild animal deaths from *A. flos-aquae* poisoning has sometimes numbered in the thousands. With the Fourth of July holiday approaching, a meeting was held to decide what protective measures should be implemented to prevent any more loss of pets, or cattle, or risks to people. The decision was to close the lake to recreational boating, and to keep the shoreline and campground on the Grayling Arm of the lake closed until the toxicity of the water had ended. Daily sampling of the Grayling Arm algae and water continued. The bloom gradually declined during July and was nontoxic by July 30th.

Possible explanations for the bloom include starting to fill the reservoir in February instead of the normal late April because of low winter snowfall and expected low snowmelt runoff that year, with subsequent early warming of the water. The 21-feet (6.4-m) drawdown of this reservoir may have allowed for bottom sediments of the Grayling Arm to be extracted for nutrients. The upper watershed lies inside Yellowstone National Park where it drains highly mineralized volcanic materials and geysers that produce a naturally high concentration of nutrients. As a result, phosphate content is relatively high. The reservoir is nitrogen limited. Juday and others (1981) classified the main part of Hebgen Lake reservoir as mesotrophic and the Grayling Arm as eutrophic. They also found the *A. flos-aquae* algae disappeared about 1 km out in the main part of the lake. Apparently the water chemistry outside Grayling Arm was inhospitable to the *Anabaena*.

If dam owners begin to fill their reservoirs earlier than normal in the spring to capture snowmelt runoff in drought years, the water has extra time to warm up. With enough nitrogen and phosphorus in the warm water from natural and manmade sources, conditions favor algae blooms. In many States, including Montana, both Dakotas, Indiana, Iowa, Minnesota, Missouri, and Wisconsin, toxic blooms of blue-green algae have been reported, even in forested and largely pristine watersheds (Carmichael 1981, Fawks and others 1994, Horpestad and others 1978). Whether a given bloom will turn toxic is still unknown. Accidental ingestion by people engaging in water sports is a risk to human health. Although no deaths have been reported, prudence calls for prohibiting all water contact sports and closure of public drinking water intakes when toxic blue-green algae blooms are suspected. Improved methods of detection of toxic blue-green algae blooms have resulted in more reports on their occurrence.

Trihalomethane

Trihalomethanes are compounds that form when chlorine or bromine, added to drinking water for disinfection, reacts with certain naturally occurring organic molecules (trihalomethanes precursors). Trihalomethanes may cause cancer and genetic mutations in humans. Researchers (Arruda and Fromm 1989, Martin and others 1993) report that reservoir and lake organic sediments can contain and release thihalomethane precursors. In one study in Ohio, all sediment samples had significantly more trihalomethane precursor releases than controls. Anaerobic conditions and deep water sediments had much fewer trihalomethane precursors than aerobic sediments from shallower zones. Karimi and Singer (1991) and Wardlaw and others (1991) reviewed the role of algae as trihalomethane precursors. They found that a variety of natural organic compounds, especially humic and fulvic acids derived from soils and decomposition of plant material, are the trihalomethane precursors. No discernible trends in the ability of particular algae species to generate trihalomethanes can be drawn from published data. Trihalomethane concentrations arising from a natural algal bloom, however, could theoretically exceed maximum allowable concentrations for drinking water. Understanding trihalomethane precursor sources is important because limiting them may lower risks to human health and lower water treatment costs. Management of a reservoir to limit algal growth may reduce water treatment costs and improve water quality in the reservoir (Kortmann 1989).

Sediments Deposited in Reservoirs

Sediment deposited in reservoirs can also pose public health problems if it contains heavy metals, radioactive elements, or pesticides and other synthetic organic compounds. Many of these chemically bond to the sediment particles under the right chemical conditions. The risk to human health often remains low as long as the sediment remains undisturbed at the bottom of a lake or reservoir. The accidental failure or deliberate removal of a dam may pose a human health problem by destabilizing accumulated sediment, but literature is lacking on this topic. Modifying streamflows has the potential to mobilize and later deposit sediment that may then reduce the quality of drinking water. See chapter 2 for more information on this topic. Further research needs to be conducted on remobilization of toxic sediments.

Splash dams and log flumes were constructed on many rivers in New England, the Lake States, and the West. The dams were earthern structures <20 feet (7 m) high with the main spillway constructed of wooden boards. They typically held from a few hundred to 1,000 acre-feet (up to 1.25 million m^3) of water. When the boards were removed, an artificial flood was created downstream, sweeping logs down the channel. Such dams are no longer constructed, and their residual effects upon drinking water quality today are likely to be minor.

Controlled removal of sediment by dredging from channels, lakes, or reservoirs can degrade domestic water supplies. These sediments pose special problems if they contain toxic substances or if they are massively released.

Water Diversion Structures and Water Import/Export Between Watersheds

Water is frequently removed from a river by means of a diversionary dam or headgate along one side of the channel. The water then enters a ditch, aqueduct, or pipeline to be carried to the place of use, often miles away. The removal of this water results in changes in the remaining river water. Concentrations of pollutants increase, water temperatures rise, and biological activity of aquatic organisms increases. The acidity of the water often rises as well, changing the solubility of metals and rates of chemical reactions in the water column. Suspended sediment transport declines as flow declines, causing increased deposition of fines on the beds of rivers (Heede 1980).

The effects of removing water from rivers upon drinking water quality at intakes located below points of diversion can usually be overcome at the water treatment plant—as long as there is enough water left to be treated. There is no scientific literature on this subject. The same is true for water added to stream channels by diversions from other watersheds or aquifers. Differences in chemical, physical, or microbiological quality of such waters may create complications when they are mixed together.

Water Well Effects on Drinking Water Quality

High pumping rates from water wells can decrease flows of nearby streams used for drinking water, sometimes for months or longer. Decreases in streamflow usually degrade drinking water quality by changing acidity, dissolved oxygen, and water temperature. Rates of pumping that exceed the recharge rate of the aquifer draw down the water table, altering the yield and water quality at other wells tapping the same aquifer.

Wells in floodplains can become contaminated during high streamflows if they are not properly protected ahead of time. Singer and others (1982) found that bacterial counts, nitrate nitrogen, turbidity, conductivity, sulfate, chloride, phosphate, total organic carbon, and several ratios of these variables were the best indicators of surface water contamination of aquifers in a karst area of southestern Minnesota. Improper sealing or grouting of the annular space of the well itself can result in cross contamination, aquifer damage, loss of well performance, and damage to the well (Ashley 1987). The most commonly used sealing materials in wells, cement and bentonite clay, have properties that can cause them to fail if unsuitable drilling and well construction methods are employed in some hydrogeologic environments. There is a large body of literature on well construction and maintenance. The reader should obtain expert assistance if it appears that local water wells could be responsible for pollutants in forest or grassland watersheds. An Internet site to go to for information on wellhead protection is EPA's Office of Ground Water and Drinking Water, located at http://www.epa.gov/OGWDW/whpnp.html.

Sewage Effluent and Sludge/Biosolids Applications to Forest and Rangeland

Return flows of sewage effluent or sludge and biosolids or both are sometimes applied to the land surface rather than returned to water bodies. Research on effluent and sludge or biosolids applications was conducted in the Pacific Northwest by Machno (1989), in New England by Koterba and others (1979), and in the Lake States and Southeastern United States by other researchers. Materials were applied under hardwood forests. Koterba and others (1979) found little change in soil water and stream chemistry after light application [11 tons per acre or 25 metric tonnes (Mg) per hectare] of limed and dewatered sludge on sandy loam soils in a northern hardwood forest in central New England. They measured short-lived and relatively small increases in calcium, magnesium, sodium, chloride, and sulfate after 56 tons per acre (125 Mg per hectare) were applied. There were no changes in infiltration capacity of the soil and no visual evidence of overland transport of the sludge.

Brockway (1988), Brockway and Urie (1983), and Sorber and Moore (1986) studied effects of applying municipal or papermill sludge and wastewater to forests by monitoring the movement of nitrogen and other constituents in the leachate and ground water. Results showed nitrate nitrogen concentrations exceeded 10 parts per million in ground water under aspen (*Alnus* spp.) plots treated once with 7 or more tons per acre (16 Mg per hectare) of undigested papermill sludge, and under pine (*Pinus* spp.) plantations receiving 8.5 tons per acre (19 Mg per hectare) per year of anaerobically digested municipal sludge in a single application. Brockway and Urie (1983) estimated that anaerobically digested municipal sludge could be safely applied to a red pine (*P. resinosa* Ait.) and white pine (*P. albicaulis* Engelm.) plantation at 7.25 dry tons per acre {880 pounds total nitrogen per acre [986 kilograms (kg) per hectare]} per year or less, and to aspen stands at rates up to 8.4 dry tons per acre [1,015 pounds total nitrogen per acre (1138 kg per hectare)] per year. Although long-term additions of nitrogen to soil could lead to nitrogen saturation (see chapter 3), this effect has not been studied for sewage sludge applications.

Spray applications of treated municipal wastewater on forests in Michigan have been studied by Urie and others (1990) and Brockway (1988). Overall, it appears that nitrate contamination of ground water can be avoided at appropriate application rates on most acidic forest soils.

Edmonds (1976) studied the survival rate over 3 years of coliform bacteria in sewage sludge applied to a forest clearcut on gravelly glacial outwash soils. Results indicated that few viable fecal coliforms penetrated deeper than 2 inches (15 centimeters) into the soil and that practically none moved into the ground water. The soil was effective as a biological filter for hazardous pathogens, but coliforms can remain viable for years in the surface soil. He concluded there was little danger of ground water contamination from vertical bacterial movement.

Harris-Pierce and others (1995) applied sewage sludge on a semiarid grassland in Colorado. They found that increasing rates of single applications from 0 to 9.7 to 18 tons per acre (22 to 40 Mg per hectare) increased concentrations of sediment, organic nitrogen, ammonia nitrogen, potassium, boron, phophorous, copper, nickel, and molybdenum in surface runoff from a single sprinkler rainfall event on the plots. All constituents remained below EPA's drinking water standards. However, Burkhardt and others (1993) argued for a careful approach to sludge applications on rangeland without irrigation because of the limited opportunity for nutrient uptake and sludge assimilation by the native vegetation. They saw risks of the nutrients and metals moving off-site when rainfall events do occur.

Sagik and others (1979) evaluated microbial survival and movement in soils subjected to sludge applications and concluded that both bacteria and viruses can survive and move through the soil profile for up to 2 years; prudence says that nondisinfected sludge should not be applied to soils used to grow crops or feed for dairy cows or livestock for human consumption. See EPA's Web site at http://www.epa.gov/owm/bio.htm for additional information about biosolids recycling.

Wetland Drainage

Effects of wetland drainage on drinking water quality have been studied. Results show some small increases in nitrogen leaching and coliform movement with the leachate, but that the drainage water is easily handled by the water treatment plant.[4]

Reclaimed Water and Return Flows

After use, water withdrawn from rivers or aquifers is often returned to these sources. Quantity and quality of the returned water may be changed, depending upon the type of use and type of treatment it receives prior to return. There is a large body of literature and regulations about sewage treatment because it is a point source of pollution under the Clean Water Act. Reuse of water effluent from sewage treatment plants is growing in the United States and has passed the 1-billion-gallon-per-day (4-billion-liter-per-day) mark for both nonpotable (water not intended for human consumption) and potable (drinkable) uses. Water reuse for nonpotable applications, such as irrigation, lawn watering, car washing, and toilet flushing is widely accepted where water supplies are scarce, as in Arizona, California, Florida, and Texas. The EPA and the National Academy of Sciences have recommended limits for many physical parameters and chemical constituents of nonpotable water. The health risks from disease-causing microorganisms are not as well known; hence, there is no direct potable reuse in the United States (Crook 1997).

Of course, indirect potable reuse occurs when effluents are treated and returned to rivers that are water sources downstream. Required treatments may include: (1) chemical clarification and two-stage recarbonation with intermediate settling, multimedia filtration; (2) activated carbon adsorption; (3) ion exchange for nitrogen removal; and (4) breakpoint chlorination. Indirect potable reuse can also occur when effluent is used for ground water recharge by means of injection wells. Some States prohibit that practice if potable aquifers would be contaminated. Other States have set stringent water-quality standards and require high levels of effluent treatment before it is returned to the aquifer (Crook 1997). Crook also lists a number of references on water reuse that would be very helpful to managers of land influenced by water reuse or officials responsible for completing source water assessments.

While irrigation return flows are exempt from the National Pollutant Discharge Elimination System permitting process, they can carry potentially harmful concentrations of pesticides, heavy metals, or other toxic substances acquired from atmospheric deposition, soils, and plants. Crop irrigation is beyond the scope of this report; there is a large amount of literature on this subject by EPA and various universities.

Reliability and Limitations of Findings

Scientific literature on the direct effects of dams, water diversion and conveyance structures, water wells, and other related engineered structures upon drinking water quality is very limited. Far more is known about their effects on physical habitats of aquatic life forms. Most of the studies mentioned did not describe how the water facility was operated or whether the manner of operation could have, or did, influence the results. Facility operational details should be better evaluated in future research studies.

The indirect effects of dams, water diversions and conveyance structures, water wells, and applications of sewage sludge should apply in all forest and rangeland watersheds in the United States. The magnitude and timing of the indirect effects will vary by region and perhaps by elevation because of variations in temperature and precipitation. None of the studies reported were national or even regional in scope, and only a few were carried out for a decade or more, so long-term trends have been ignored or are not known.

[4] Personal communication. 1999. James D. Gregory, Professor of Watershed Hydrology, Department of Forestry, North Carolina State University, Raleigh, NC 27695.

Research Needs

1. The direct effects of dams and their operation, mobilization of sediments when dams are removed, water diversion and conveyance structures, and water wells upon human drinking water quality need to be studied.
2. Research is needed to determine why some blue-green algae blooms turn toxic and how to predict the toxicity levels.

Key Points

1. Managers who experience blue-green algae blooms in their reservoirs need to recognize that such blooms sometimes become toxic without prior warning or previous history. These toxins are invisible when released by the algae into the water, and are extremely deadly to all mammals if ingested. Most of the other risks to human health from water storage and control structures are known and can be assessed in local watersheds by professionals in hydrology and health sciences.
2. Risks from applying sewage sludge on forest and rangelands are manageable as long as disease-causing organisms have been killed at the sewage treatment plant before the sludge is applied to the soil.
3. Improper construction or inadequate well head protection of water wells can be a cause of ground water contamination. People doing source water assessments in forest and rangeland watersheds should carefully examine wells in the vulnerability assessment.

Literature Cited

Arruda, J.A.; Fromm, C.H. 1989. Relationships among trihalomethane formation potential, organic carbon and lake enrichment. Environmental Pollution ENPOEK. 61(3): 199–209.

Ashley, J.W. 1987. Use and placement of grouting and sealing materials. In: Proceedings of the 4th annual eastern regional ground water conference; 1987 July 14–16; Burlington, VT. Dublin, OH: National Water Well Association: 735–748.

Brockway, D.G. 1988. Forest land application of municipal sludge. Biocycle. 29(9): 62–68.

Brockway, D.G.; Urie, D.H. 1983. Determining sludge fertilization rates for forests from nitrate-N in leachate and ground water. Journal of Environmental Quality. 12(4): 487–492.

Burkhardt, J.W.; Miller, W.W.; Azad, M. 1993. Biosolids application to rangelands. Water Environment and Technology. 5(5): 68–71.

Carmichael, W.W. 1981. Freshwater blue-green algae (cyanobacteria) toxins—a review. In: Carmichael, W.W., ed. The water environment: algal toxins and health.: Proceedings of an international conference on toxic algae; 1980 June 29–July 2; Dayton, OH. New York: Plenum Press. 491 p.

Crook, J. 1997. Water reuse comes of age. In: Proceedings of water resources management: preparing for the 21st century; 1997 August 10–13; Seattle. Denver: American Water Works Association: 487–501.

Edmonds, R.L. 1976. Survival of coliform bacteria in sewage sludge applied to a forest clearcut and potential movement into ground water. Applied and Environmental Microbiology. 32(4): 537–546.

Fawks, D.J.; Kotak, B.G.; Jones, J.R.; Prepas, E.E. 1994. Occurrence of hepatotoxic blue-green algae in Missouri reservoirs. Lake and Reservoir Management. 9(2): 72.

Getches, D.H.; MacDonnell, L.J.; Rice, T.A. 1991. Controlling water use—the unfinished business of water quality protection. Boulder, CO: University of Colorado School of Law, Natural Resources Law Center. 12 p.

Harris-Pierce, R.L.; Redente, E.F.; Barbarick, K.A. 1995. Sewage sludge application effects on runoff water quality in a semiarid grassland. Journal of Environmental Quality. 24(1): 112–115.

Heede, B.H. 1980. Stream dynamics: an overview for managers. Gen. Tech. Rep. RM–72. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 26 p.

Horpestad, Abe; Bahls, L.L.; Glasser, S.P. 1978. Green creatures of the deep—the Hebgen Lake algae bloom of 1977. Montana Science. 1(1): 33 p.

Juday, R.E.; Keller, E.J.; Horpestad, A. [and others]. 1981. A toxic bloom of *anabena flos-aquae* in Hebgen Lake Reservoir, Montana, in 1977. In: Carmichael, W.W., ed. The water environment: algal toxins and health. New York: Plenum Press. 491 p.

Karimi, A.A.; Singer, P.C. 1991. Trihalomethane formation in open reservoirs. Journal of the American Water Works Association. 83(3): 84–88.

Kortmann, R.W. 1989. Raw water quality control: an overview of reservoir management techniques. Journal of the New England Water Works Association. 103(4): 197–220.

Koterba, M.T.; Hornbeck, J.W.; Pierce, R.S. 1979. Effects of sludge applications on soil-water solutions and vegetation in a northern hardwood stand. Journal of Environmental Quality. 8(1): 72–78.

Machno, P.S. 1989. Seattle's success with silvigrow broadens sludge management options. In: Proceedings of the national conference; 1989 May 27–29; Boston. Sludge Manage. Ser. 17. Silver Spring, MD: Hazardous Materials Control Research Institute: 131–134.

Martin, A.B.; Cooke, G.D.; Carlson, R.E. 1993. Lake sediments as potential sources of trihalomethane precursors. Water Resources Research. 27(12): 1725–1729.

Reetz, G.R.; Berkley, J.; Campbell, D. [and others]. 1998. Water quality in the West. Report to the western water policy review advisory commission. Denver: U.S. Environmental Protection Agency. 183 p. + appendices.

Sagik, B.P.; Moore, B.E.; Sorber, C.A. 1979. Public health aspects related to the land application of municipal sewage effluents and sludges. In: Sopper, W.E.; Kerr, S.N., eds. Utilization of municipal sewage effluent and sludge on forest and disturbed land. University Park, PA: The Pennsylvania State University Press: 241–253.

Singer, R.D.; Osterholm, M.T.; Straub, C.P. 1982. Ground water quality in southeastern Minnesota. Water Resour. Res. Cent. Bull. 109. St. Paul, MN: University of Minnesota. 45 p. Available from the National Technical Information Service, 5285 Port Royal Road, Springfield, VA 22161; PB83–15437.

Sorber, C.A.; Moore, B.E. 1986. Microbiological aspects of forest application of wastewater and sludge. In: Cole, D.W.; Henry, C.L.; Nutter, W.L., eds. The forest alternative for treatment and utilization of municipal and industrial wastes. Seattle: University of Washington Press: 73–84.

Urie, D.H.; Hart, J.B.; Burton, A.J. 1990. Nitrification in sludge—amended Michigan forest soils. Journal of Environmental Quality. 19(3): 609–616.

U.S. EPA. 1995. National water quality inventory: 1994 report to Congress. EPA–841–R–95–005. Washington, DC: U.S. Environmental Protection Agency. 497 p.

Wardlaw, V.E.; Perry, R.; Graham, N.J.D. 1991. The role of algae as trihalomethane precursors: a review. Aqua. 40(6): 335–345.

Chapter 6

Urbanization

Wayne C. Zipperer, Karen Solari, and Beverly A. Young[1]

Introduction

This chapter specifically examines drinking water issues related to urbanization. The discussion is limited principally to land that is developed or being developed within and adjacent to public land. Urbanization issues include current and past land uses and Forest Service facilities. Forest Service buildings and administrative sites are included because they are similar to other developed sites.

The effects of urbanization on drinking water quality encompass many topics with extensive published literature. Because of limited space, selected topics of current and past land use are examined including wastewater treatment, urban storm runoff, underground storage tanks, abandoned wells, and landfills. However, land managers need to realize that these selected topics do not include all effects caused by urbanization, land fills, and abandoned mines on drinking water and serve only to illustrate potential effects.

Vitousek (1994) has identified land cover changes as a primary effect of humans on natural systems. With the projected global increase in urbanization, land cover conversions for urban use will only increase. In this chapter, we examine the potential impacts on drinking water of current and past use of land in and adjacent to public land. Nationally, development and growth rates are not available for such land. To estimate these rates, we have used in-holding data from the Forest Service.

Inside the boundaries of publicly owned land are parcels not administered by the Agency. They are called in-holdings. In-holdings are managed or owned by other Federal, State, local, and tribal government agencies, and by private landowners. Of particular interest is the private land because of its propensity for development. The occurrence of in-holdings varies by Forest Service region (table 6.1). The States that comprise each region are listed in table 6.1.

Region 4 has the least area of in-holdings; only 6.9 percent of the land inside national forest boundaries. By comparison, Regions 8 and 9 had 48.6- and 45.6-percent in-holdings within national forest boundaries, respectively.

Unfortunately, no data are available on how rapidly these in-holdings are being developed. To estimate this rate, we used the growth rates of counties that intersect with or are adjacent to a national forest. Population growth was calculated for 1980–90 and 1990–96 using census data (U.S. Census Bureau 1997). Between 1980 and 1990, the population of these counties grew by 18.5 percent, while the Nation's population grew by 9.8 percent (table 6.1). For 1990 to 1996, the population in these counties grew by 10.1 percent, while the Nation's population grew by 6.4 percent. In 1996, these counties contained 22.4 percent of the Nation's population (U.S. Census Bureau 1997).

Population change in these counties varied by region and time period. Between 1980 and 1990, Region 5 experienced the greatest percent increase (28.9 percent), while Region 1 experienced a decrease of 3.5 percent. Between 1990 and 1996, Region 4 and 6 showed the largest increases of 14.9 and 14.3 percent, respectively, and Region 9 showed the least growth of 5.2 percent. Overall, Regions 2, 3, 4, 5, and 6 had growth rates greater than the national average for the period between 1990 and 1996. The effects of this development on drinking water quality depend on the location within the watershed, the concentration of development, and existing conditions. Unfortunately, data are unavailable to examine those variables.

Issues and Risks

During the past 20 years, private tracts in and adjacent to public land have been developed rapidly for residential, commercial, and recreational use. This development poses a significant threat to drinking water quality through surface and ground water contamination. Development occurs near the headwaters of streams where water quality is generally the highest and is easily degraded because of stream size. For ground water, about 95 percent of rural communities use

[1] Research Forester, USDA Forest Service, Northeastern Forest Research Station, Syracuse, NY; Environmental Engineer, USDA Forest Service, Washington, DC; Environmental Engineer, USDA Forest Service, Northern Region, Missoula, MT, respectively.

Table 6.1—Reported acres inside national forest boundaries and percent population changes in counties containing or adjacent to national forests by administrative Forest Service Region in the conterminous United States

Region[a]	Area[b]				Population change 1980–90	Population change 1990–96
	Gross	NF	In-holding	Other		
	Acres				*Percent*	
1	28,180,534	25,375,333	2,805,201	10.0	-3.5	5.7
2	24,477,648	22,098,044	2,379,604	9.7	3.6	10.6
3	22,381,905	20,702,312	1,679,593	7.5	17.2	10.9
4	34,257,094	31,903,934	2,353,160	6.9	14.8	14.9
5	23,739,894	20,022,650	3,717,244	15.7	28.9	10.3
6	27,357,569	24,629,048	2,728,521	10.0	6.8	14.3
8	25,034,868	12,874,851	12,160,017	48.6	6.1	6.8
9	21,934,418	11,942,218	9,992,200	45.6	1.8	5.2
Total	207,363,930	169,548,390	37,815,540	18.2[c]	9.8[d]	6.4[d]

[a] Region 1: Montana, northern Idaho, North Dakota, and northwestern South Dakota; Region 2: Colorado, Kansas, Nebraska, and southeastern South Dakota; Region 3: Arizona and New Mexico; Region 4: southern Idaho, Nevada, Utah, and western Wyoming; Region 5: California, Hawaii, Guam, and Trust Territories of the Pacific Islands; Region 6: Oregon and Washington; Region 8: Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, Puerto Rico, South Carolina, Tennessee, Texas, and Virginia; and Region 9: Connecticut, Delaware, Illinois, Indiana, Iowa, Maine, Maryland, Massachusetts, Michigan, Minnesota, Missouri, New Hampshire, New Jersey, New York, Ohio, Pennsylvania, Rhode Island, Vermont, West Virginia, Virgin Islands, and Wisconsin.
[b] In-holding = parcels of land within the boundaries of publicly owned lands that are not administered by the public agency; NF = national forest; Other = percent of in-holdings within the gross acreage inside national forest boundaries.
[c] Percent of in-holdings within national forest boundaries for the conterminous United States (calculated as in-holdings/gross).
[d] Percent population change in counties containing or adjacent to national forests for the conterminous United States.

ground water as the principal source of drinking water. Sources of pollution result from wastewater treatment, nonpoint-source pollution, underground storage tanks, solid waste storage, and hazardous material storage. The extent of ground water contamination depends on depth of ground water. Shallow ground water sources < 100 feet [30 meters (m)] below land surface may be more readily and significantly contaminated than deeper ground water sources (U.S. Geological Survey 1999).

In 1995, the U.S. Environmental Protection Agency (EPA) (U.S. EPA 1998d) summarized water-quality information submitted by States, tribes, and other jurisdictions. For rivers, streams, lakes, ponds, and reservoirs, municipal point and nonpoint sources from residential and commercial sources were identified as significant contributors of pollution to rivers, streams, lakes, ponds, and reservoirs. For ground water, principal sources included leachate from leaking underground storage tanks, septic tanks, and landfills. Of specific importance is the effect of urbanization on the quality of surface water and ground water in rural areas (table 6.2).

The report identified urbanization as a major factor in contaminating surface and ground water, and modifying hydrologic processes. Urbanization replaces natural vegetation cover with impervious surfaces, decreasing natural infiltration of water, increasing peak flows, and decreasing ground water recharge (Weiss 1995). Increased peak flows can negatively affect drinking water quality by causing bank destabilization and streambed scouring, which increase turbidity and sedimentation (Phillips and Lewis 1995). Reduced ground water recharge decreases baseflow in streams and increases pollutant concentrations. Decreased baseflow impairs aquatic habitat and riparian wetlands and increases the stream's sensitivity to pollution and sedimentation (Weiss 1995).

Table 6.2—Estimated use from freshwater surface and ground water sources in the United States, 1980–95

Source	1980	1985	1990	1995
	Cubic kilometers			
Ground	120	101	110	105
Surface	400	366	358	364
Total	520	467	468	469

Source: Adapted from Gleick 1999.

Wastewater Treatment

Residential and commercial wastewater is treated by decentralized and centralized systems. Decentralized systems treat water onsite. They include individual and large septic systems, and cluster wastewater systems. Generally, septic systems treat and dispose of relatively small volumes of wastewater. They are for individual dwellings or groups of dwellings and businesses located close together. A centralized system is a collection and treatment system containing collection sewers and a central treatment facility (U.S. EPA 1997a). Centralized systems are used to collect and treat large volumes of water. Decentralized systems affect both surface and ground water, while centralized systems generally affect surface waters.

The 1990 census indicates that 25 million households use onsite disposal systems for wastewater. Data on the failure rates associated with these systems are limited and no national estimates are available. Each State has its own definition of failure, but estimates of failure rates range from 18 to over 70 percent (U.S. EPA 1997a). Twenty-seven States have cited onsite disposal systems as a potential source of ground water contamination (U.S. EPA 1998d). Contaminants from onsite disposal can be classed as inorganic (sodium, chlorides, potassium, calcium, magnesium, sulfates, and ammonium), microorganisms (bacteria and viruses), and chemical organics originating in household products (Phillips and Lewis 1995, U.S. EPA 1997a). Effluent from septic systems usually contains high concentrations of ammonium and organic nitrogen.

Water supplies are vulnerable to pathogenic bacteria and viruses from onsite disposal systems. Reported outbreaks of waterborne disease in the United States are uncommon (table 6.3), but 404,000 people fell ill to a *Cryptosporidium* spp. outbreak in Milwaukee, WI, in 1993. To some extent, low occurrence may be attributed to individuals being unaware that their illness was a waterborne disease or to the number of illnesses being so small that they go unreported by local health departments. Ground water sources have a higher incidence of waterborne outbreaks than surface water because ground water often is not filtered or disinfected before it is used for drinking (table 6.4). Disease-causing microorganisms isolated from domestic sewage include *Salmonella*, *Shigella*, pseudomonas, fecal coliform, and protozoa (*Giardia lamblia*) (U.S. EPA 1997a). Other microorganisms found in contaminated drinking water include *Cryptosporidum*, *Microsporidum*, *Cyclosporidium*, *Helicobacter pylori*, hepatitis E, and the enteric viruses hepatitis A and Norwalk virus (U.S. EPA 1997b). See chapter 2 for a more thorough discussion of waterborne pathogens.

Table 6.3—Waterborne disease outbreaks in the United States by water supply system, 1990–94

Source	1990	1991	1992	1993	1994
Municipal	5	2	9	9	5
Semi-public	7	13	14	4	5
Individual	2	0	4	5	2
Total outbreaks	14	15	27	18	12
Total cases	1,758	12,960	4,724	404,190[a]	1,176

[a] Includes Milwaukee, WI.
Source: Adapted from Gleick 1999.

Table 6.4—Comparison of outbreak percentages by drinking water source from pathogenic contamination for the period 1971–96[a]

Water source	Total outbreaks		Cases of illnesses	
	No.	*%*	*No.*	*%*
Ground	371	58	84,408	52
Surface	215	33	66,721	41
Other	56	9	10,625	7

[a] Excludes outbreak in Milwaukee, WI, 1993.
Source: Adapted from Craun and Calderon 1996, U.S. EPA 1999.

The fate and transport of parasites, bacteria and viruses from sewage effluent depend on the characteristics of the subsurface environment (U.S. EPA 1997a). Pore size and chemical charges of the soil matrix are important in removing bacteria and viruses. Bacteria have been reported to travel distances of up to 300 feet (100 m) in sand aquifers, 2,500 feet (800 m) in gravel aquifers, and over 3,000 feet (1000 m) in limestone rock (Kaplan 1991). Certain viruses, because of their size and long survival times, can travel distances up to 1 mile [1.6 kilometers (km)] in areas with karst geology (Yates and Yates 1989).

Organic chemicals in onsite disposal systems are a less commonly reported problem because they often are below levels considered hazardous to human health (U.S. EPA 1997a). These chemicals can significantly affect aquatic systems, however. Organic chemicals commonly found in septic systems originate from household products, paints and varnishes, shampoos, cosmetics, and polishes.

Septic systems fail for two reasons: poor design or poor maintenance (Kelley and Phillips 1995). Design includes construction, soils and hydrological characteristics of the site, and drainfield layout (Kelley and Phillips 1995). If drainage is too slow, there will be upward seepage and ponding, which are likely to contaminate surface water. If drainage is too fast, downward percolation occurs without sufficient biological treatment; contamination of ground water is likely to result.

Even with properly installed systems, maintenance is absolutely necessary. Unfortunately, the typical owner of an onsite disposal system is unaware of the need for proper maintenance (Kelley and Phillips 1995). Maintenance includes periodic testing of drainfields and emptying of septic tanks. Frequency of maintenance depends on soil conditions, type of septic system, and weather patterns.

Class V injection wells is another type of onsite disposal unit. An injection well can include any manmade hole in the ground for injection of wastewater (U.S. EPA 1998a). They are used by dry cleaners, laundromats, paint dealers, hardware stores, funeral homes, and other industrial and commercial facilities for materials other than domestic and sanitary wastes. Motor vehicle waste disposal wells, industrial waste disposal wells, and large-capacity cesspools have high risk for ground water contamination. Field studies have shown that ground water sources can be degraded significantly by organic and inorganic contaminants from dry wells in automotive shops (Ogden and others 1991). See the section on abandoned wells in this chapter.

Approximately 10 percent of the wastewater produced in the United States originates from communities of $<$10,000 people. With the passage of the Clean Water Act in 1972, many such communities elected to use Federal funds to install centralized systems for wastewater treatment. In small communities, contractors frequently installed the most economical and not necessarily the most effective systems. Currently, many of these systems are obsolete and need replacing because they have operated beyond their 20–year life span. Small communities also face an economic factor of scale. Costs of maintenance and staffing must be divided among fewer people, resulting in higher costs per person. Consequently, small communities have nearly twice the number of violations than larger communities ($>$10,000 individuals). Violations include leaking sewage systems (cracked and broken sewer lines), illegal connections of sewer and storm drainage lines, and inadequate treatment. Violations often affect local water quality and potentially affect drinking water quality for downstream communities. Since 1970, new technologies have been developed to treat water more effectively and cheaply. However, many small communities have not adopted these systems because of a lack of knowledge, public distrust of new technologies, and legislative and regulatory constraints (U.S. EPA 1994). Additional discussion of centralized wastewater treatment can be found in chapter 5.

Urban Runoff

Findings

Urban land generates nonpoint-source pollution. People apply various chemicals around their homes, businesses, and adjacent land. These chemicals are carried by surface runoff to receiving waters. As land is developed and impervious surface area increases, the amount of urban runoff increases. Consequently, land development increases the amount of nonpoint-source pollutants discharged into surface water (Phillips and Lewis 1995). The Nationwide Urban Runoff Program (U.S. EPA 1983) reported that 10 times as much suspended solid material was being discharged from storm sewers serving residential and commercial areas as was discharged from sewage treatment plants providing secondary treatment (Weiss 1995). Major pollutants associated with residential and commercial runoff include floatables, sediments, suspended solids, oxygen-demanding materials, nutrients, organics, biocides (herbicides, fungicides, pesticides), polycyclic aromatic hydrocarbons, and petroleum hydrocarbons (U.S. EPA 1997a, Weiss 1995).

Because residential and commercial construction creates site disturbances, it is highlighted here. Sediment loading from site preparation, and construction and maintenance of buildings and roads can exceed the capacity of streams to transport it (Yoder 1995). Sediment loads from inadequately controlled construction sites typically are 10 to 20 times per unit of land area those from agricultural land and 1,000 to 2,000 times those from forest (Weiss 1995). In a relatively short period, urban site construction can contribute more sediment to a stream than was deposited over the previous several decades (Weiss 1995).

Urban runoff is highly intermittent. Short-term loading, associated with individual storms, is high and may have a shock effect on the receiving water (Weiss 1995). When predicting the effect of urban runoff on water quality, it is important to determine the duration of the effect. Effects may be acute (short term) or chronic (long term) (Phillips and Lewis 1995). Oxygen-demanding substances and bacteria create acute effects; whereas nutrients, sediments, toxic metals, and organics create chronic effects. For an acute effect, estimates are based on the probability that pollution concentrations will exceed acceptable drinking water standards (Phillips and Lewis 1995). For a chronic effect, a simple method has been developed to predict the increase in pollution loading above current conditions (U.S. EPA 1983). This simple method is employed by EPA and uses information readily available to the resource manager. Input variables include pollutant type and concentration, precipitation, and percent impervious cover. The method, however, is limited to areas <1 square mile (2.6 km^2).

Findings from engineering research show that pollution and sediment loading from runoff can be reduced. Practices for mitigating storm runoff include attenuation, conveyance, pretreatment, and treatment of runoff (U.S. EPA 1997a). When selecting mitigation practices, it is important to consider

- How will practices meet watershed and site objectives?
- What are the limitations of a practice to meet objectives?
- What are the drainage field, soil types, and topography?
- Are practices compatible with a region’s rainfall pattern and annual runoff?
- Are they derived from scientific research?
- How will practices function as a system (Phillips and Lewis 1995)?

A number of manuals and practical guides have been written to select, design, and maintain mitigation practices to meet local, State, and Federal mandates (Birch 1995, Phillips and Lewis 1995). As with plans and guides for wastewater treatment, managers need to check with State and local agencies for specific performance ratings and regulations. Like wastewater treatment facilities, new mitigation practices must be maintained and existing ones upgraded to meet expected performance standards. Adequate funds often are lacking to maintain or enhance these facilities (U.S. EPA 1997a). Without proper maintenance, water quality degrades as systems fail.

Reliability and Limitation of Findings

Although development of private land in or adjacent to public land has occurred for decades, scientific studies of the effects on water quality and drinking water sources are lacking. Extensive research has been conducted on urban effects on natural systems, however. These studies provide the basis for identifying the potential impacts of development on drinking water.

When applying findings across a watershed, scale becomes an important issue. Evaluating cumulative effects requires examination of more than just local impacts of individual pollution sources, such as urban runoff, wastewater treatment, and landfills. The timing and location of all activities that contribute contaminants within the watershed and their hydrologic connection to source water intakes must be considered to estimate cumulative effects. Consequently, these developments must be evaluated both independently and collectively within the watershed. Ages of wastewater treatment facilities and urban storm runoff structures must be considered. For various reasons, existing infrastructures may not meet sanitation and water-quality regulations. Success of management plans to mitigate the effects of wastewater treatment and urban runoff is predicated on sound infrastructure.

The ability to address the effects of development on drinking water quality depends on ownership. On publicly owned land, resource managers directly determine whether facilities comply with Federal and State regulations. On privately owned land, resource managers can only indirectly influence development effects on drinking water quality through the planning process.

Planning and development of private land in and adjacent to public land involve complex issues including the interplay of the physical, biological, and social components of a watershed. A number of factors need to be considered. First, planning must include all stakeholders, including public land managers. Second, private tracts are owned by a diversity of individuals for various reasons. Third, new regulations often cause resentment among landowners. Any changes in drinking water regulations and statutes create the need for communication and education. Fourth, a comprehensive approach is needed to account for the cumulative effects of individual developments in a watershed and to address the needs of individual stakeholders.

Because of the interplay, technology and management practices are not the only solutions to drinking water issues. A number of communities have adopted a whole watershed approach to manage water and land planning issues (Birch 1995, Kelley and Phillips 1995, Phillips and Lewis 1995). This approach provides a framework not only to design the optimal mix of water-quality management strategies but also to design land management strategies by integrating and coordinating management priorities across stakeholders, governments, and agencies. Livingston (1993) identifies the big "C's" of watershed management that must be considered:

- Comprehensive management.
- Continuity of management over a long period.
- Cooperation among Federal, State, local, and tribal governments; cities and counties; public and private sectors; and all citizens.
- Communication to educate elected officials and ourselves.
- Creativity in best-management-practice technology.
- Coordination of stormwater retrofitting to reduce pollution loading.
- Consistency in implementing laws, rules, and programs.
- Commitment to solving current problems and preventing future ones.
- Cash in funding programs and maintenance over a long period.

Research Needs

Development of private tracts in and adjacent to public land represents an opportunity to examine how development alters ecosystem processes and what are the long-term implications of these changes.

1. Long-term monitoring stations are needed not only to monitor changes in water quality and habitat modification but also atmospheric deposition.
2. In addition, studies are needed to determine the limitation of management practices, wastewater treatment, and urban runoff in extreme environments such as at high elevations [>8,000 feet (>2,400 m)]. Research also is needed to determine threshold levels of the corresponding changes in processes that affect source water quality as land use shifts to urban.

Key Points

1. Levels of drinking water protection need to increase with increasing amounts of urban development.
2. Because of their depth, shallow ground water sources are especially prone to contamination from septic systems.
3. Septic systems fail for two reasons: poor design and poor maintenance.
4. Septic system designs need to consider site conditions, such as soil characteristics (permeability, depth to bedrock, depth to ground water table), topography (floodplain, hillslope, ridge top), and climatic patterns (rainfall and snowfall amounts and patterns, winter temperatures).
5. A comprehensive approach towards development planning must be taken. The approach needs to consider issues ranging from the local to watershed scale.
6. Urban runoff is reduced by maintaining and enhancing existing vegetation and by minimizing the amount of impervious surfaces.

Underground Storage Tanks

Issues and Risks

Underground storage tanks pose a risk of ground water contamination because nearly all tanks contain petroleum products. The tanks are associated with service stations, convenience stores, and organizations that have fleets of vehicles (U.S. EPA 1998b). Current estimates indicate that 25 to 35 percent of these tanks do not comply with existing regulations. In 1986, EPA published regulations with the goals of preventing and cleaning up releases from underground storage tanks. These regulations (40 CFR 280) require that underground storage tanks, which contain hazardous substances, including fuels, be removed by December 1998 or have spill, overfill, and corrosion protection. The regulations also require that installation and closure of underground storage tanks must be registered with the State or EPA. These regulations have had a significant impact on land management agencies, which, due to the remote locations of administrative offices, recreation sites, and workshops, have installed underground storage tanks for easy access to fuel. For example, in order to comply with these regulations, the Forest Service has removed over 1,600 underground storage tanks and has initiated several projects to cleanup contaminated soil caused by leaking tanks.

The primary concern about underground storage tanks is leakage, which can seep into the soil and contaminate ground water. Since 1988, over 330,000 confirmed releases have occurred from regulated underground storage tanks. Gasoline is the most common contaminate of ground water. Although not all of those releases contaminated ground water, drinking water wells have been shut down because of petroleum contamination (U.S. EPA 1996). In 1988, EPA regulations established minimum standards for new tanks and required owners to upgrade existing tanks, to replace them, or close them by December 1998 (U.S. EPA 1996).

Findings

Recent studies have identified methyl tertiary butyl ester (MTBE) as a potential major health hazard in drinking water. Methyl tertiary butyl ester is added to gasoline to increase its oxygen content and to reduce airborne emissions. Effects on drinking water include widespread impacts from low concentrations and local impacts from high concentrations (U.S. EPA 1998c) (see chapter 7 for more detailed information on the effect of vehicular emissions).

Local impacts primarily result from leaking underground storage tanks. A survey of ground water plume data from over 700 service stations showed that 43 percent of the sites had MTBE concentrations >1,000 micrograms (µg) per liter. However, a survey of drinking water wells from 20 National Water Quality Assessment study units showed that 2 percent of 949 rural wells had a median concentration of approximately 0.5 µg per liter (well below the EPA drinking water advisory of 20 to 24 µg per liter) (U.S. EPA 1998c, Zogorski and others 1998). A study of private wells in Maine showed 1.1 percent of 951 wells with MTBE levels exceeding 35 µg per liter. Maine officials estimated that 1,400 to 5,200 private wells across the State could be contaminated at levels exceeding 35 µg per liter (U.S. EPA 1998c). The potential threat of underground storage tanks contaminating ground water should diminish as older tanks are upgraded and sites are cleared of contaminates.

Reliability and Limitation of Findings

Records should be available through the State or EPA identifying where underground storage tanks are located, where cleanup operations are ongoing, and where tanks have been removed. The possibility also exists that underground storage tanks may be present and not registered with the appropriate agency. During field visits, resource managers need to look for indications of former structures or operations on the property, and they need to note the presence of partially exposed, capped, or uncapped pipes. These pipes may be vent pipes or fill pipes for underground storage tanks. On properties where motor vehicles were operated regularly, be skeptical where there is no apparent refueling source. An underground storage tank is likely to be present (U.S. Department of Agriculture, Forest Service 1999).

Research Needs

1. More data are needed to determine the extent of contamination of drinking water sources by MTBE and the potential health hazard.
2. Research also is needed to develop more effective and cost efficient cleanup methods. Cleanup of ground water and soil contaminated by leaking underground storage tanks can be expensive and take long periods of time.

Key Points

Underground storage tanks are a potential threat to drinking water supplies through contamination of surface and ground water by storage tanks that have leaked or have been overfilled.

Abandoned Wells

Issues and Risks

Abandoned wells and wells that are no longer used may or may not have been properly closed or plugged after their use ceased. Abandoned wells are of concern because they can serve as conduits for migration of contaminants into aquifers and between aquifers.

Numerous types of abandoned wells exist on public land. Some were drilled for mineral exploration, others for oil and gas production, and still others for stock watering. Those associated with administrative and recreational developments include water wells for irrigation and drinking water and disposal wells for stormwater runoff or waste products from vehicle shops. Septic systems may be considered disposal wells when industrial or commercial wastes are treated along with sanitary wastes.

Although Federal, State, and local regulations address proper closure of abandoned wells, not all abandoned wells have been closed or plugged properly. Many of the improperly closed wells were abandoned before regulations existed. Other wells have been abandoned temporarily to allow for further use if the need should arise. Certain wells, such as automotive dry wells in vehicle shops, may still be in use but would be banned or subject to permit under proposed regulations for Underground Injection Control (U.S. EPA 1998e). Certain States already have banned such dry wells and have required cleanup, per the Resource Conservation and Recovery Act of 1976, due to contamination at such sites.

The number of abandoned wells on public land is unknown. For example, the Forest Service has inventories of some categories of in-use wells but not of abandoned wells. Knowledge of number and location of such wells is limited, and in most instances, might be gained only by a field survey.

Findings

Abandoned wells are commonly cited as avenues of contamination in Federal, State, and local programs dealing with ground water protection (Nye 1987). The EPA's Adopt Your Watershed campaign supports properly closing abandoned wells. Many States, such as Iowa, Kansas, and Nebraska, provide financial incentives for proper well abandonment because it is considered so important for ground water protection.

Proper abandonment of water wells is regulated at the State or local level. Oil and gas well closure is specified in 43 CFR 3160. Motor vehicle waste disposal wells (dry wells) are regulated in the underground injection control program as class V wells (40 CFR 146).

Field studies have shown that ground water sources can be degraded significantly by organic and inorganic contaminants from stormwater runoff and dry wells in automotive shops (Ogden and others 1991). In certain geologic formations, abandoned water wells are prone to collapse, and, when wells are drilled through multiple aquifers, contamination problems may occur (Blomquist 1984). Gass (1988) reported that abandoned water supply wells became conduits for cross contamination between aquifers. Abandoned oil and gas wells allowed leakage of contaminated or highly mineralized water, leading to ground water pollution including salinization (Gass 1988). Even plugged boreholes may have defects in structural integrity, allowing pollutant transport between confined aquifers (Avci 1992).

Reliability and Limitation of Findings

The issue of abandoned well closure is well defined in Federal, State, and local regulations. The extent of the problem on public land is unknown because wells have not been inventoried. Proper well closure is heavily regulated at present, but not heavily enforced. Existence of improperly closed wells does not mean ground water contamination will occur; only that it is has the potential to occur.

Wells on public land possess the same general characteristics as other abandoned wells. Drilling and development methods for all types of wells have usually followed industry standards. For all types of wells, the newer the well the more likely that it was drilled and closed properly. On public land, dry wells in vehicle shops may not have as much waste or as much variety of waste in them as a commercial facility would, but the pollution potential still exists. Some could have greater potential for contamination than others because of hydrogeologic formations and duration of well use. For example, in the Allegheny and Appalachian Mountains, where abandoned oil and gas wells are more numerous and older, problems may be greater than in other regions of the country.

Research Needs

1. Methods need to be developed to inventory abandoned wells on both public and private land. Inventorying methods need to incorporate the capabilities of remote sensing technology and Geographical Information Systems.
2. The inventorying process also needs to be linked to a monitoring program.
3. Further, an abandoned-well typology needs to be developed that integrates type of well, geological formation, soil, typography, climate, and potential for ground water contamination.

Key Points

1. Abandoned wells may serve as conduits for the transport of pollutants.
2. Where there may be no records of abandoned wells on a property, the property must be surveyed to locate wells.
3. The type of abandoned well influences the types of pollution that may enter ground water sources.
4. Improperly sealed abandoned wells may be a source of contamination.

Solid Waste Landfills and Other Past Land Uses

In 1990, citizens in the United States generated over 195 million tons [215 million metric tonnes (Mg)] of municipal solid waste. Currently, over 6,000 regulated municipal landfills exist (U.S. EPA 1993). However, an estimated 30,000 to 50,000 unregulated waste disposal sites are thought to exist in the United States (Woldt and others 1998). Both regulated and unregulated sites may have impacts on water quality and the environment. In 1976, the Resource Conservation and Recovery Act (RCRA) addressed waste management and separated hazardous waste management from solid waste management. Prior to RCRA, municipal and industrial wastes were deposited at the same landfills. The practice was to spread hazardous waste sludge and liquids over municipal waste, using the municipal waste to soak up the sludge (Brown and Donnelly 1988). Consequently, landfills existing prior to RCRA may contain hazardous waste and may be the origin of organic compounds found in municipal landfill leachate. Other sources of hazardous materials in landfills include household and agricultural materials, incinerator ash, and sewage sludge.

The U.S. EPA (1993) defines a municipal solid waste landfill as:

> A discrete area of land or an excavation that receives household waste, and that is not a land application unit, surface impoundment, injection well, or waste pile, as those terms are defined in the law. (Household waste includes any solid waste including garbage, trash, and septic waste derived from houses, apartments, hotels, motels, campgrounds, and picnic grounds.) A municipal solid waste landfill unit also may receive other types of waste such as commercial solid waste, non-hazardous sludge, small quantities of generator waste, and industrial solid waste.

In many rural areas, small communities are served by small landfills that may be exempt from some regulatory requirements. The U.S. EPA (1993) defines a small landfill as one that receives less than an average of 20 tons (22 Mg) of waste per day, receives <25 inches (62.5 centimeters) of rain per year, and shows no evidence of ground water contamination. About half of the regulated landfills serve communities with <10,000 people and are considered small landfills. Many of these small landfills may be on or adjacent to public land.

Issues and Risks

Municipal solid waste landfills that contaminated ground water often were poorly designed, located in geologically unsound areas, or accepted toxic materials without proper safeguards (U.S. EPA 1993). Decomposing municipal solid waste in landfills form leachates, liquids containing extremely high concentrations of organic and inorganic pollutants. Ground water contamination is common near landfills, but the effect may decrease with distance (Borden and Yanoschak 1990). A study of 71 North Carolina sanitary landfills found that 53 percent had ground water violations for organic and inorganic pollution based on North Carolina ground water-quality standards (Borden and Yanoschak 1990). Only a few landfills had organic contamination. When predicting the performance of a landfill, it is important to know its age, history of material disposal, design, and capability of handling toxic waste.

Another threat of landfills to ground water is volatile organic compounds (VOC). Volatile organic compounds come from biological and chemical degradation of materials in the landfill. Recently, VOC's have been detected in ground water (Baker 1998) and management procedures have been developed to minimize this threat (Rickabaugh and Kinman 1993). Ground water contamination was linked to methane diffusion as VOC concentrations increased. Mitigation involves improving gas removal systems at the landfill (Baker 1998). The extent of ground water contamination by VOC's and subsequent health effects need to be evaluated further.

Illegal dumping may occur on or adjacent to public lands. This practice is usually done to avoid disposal fees or the time and effort required for proper disposal. Dumped materials may include nonhazardous material such as scrap tires, yard waste, and construction waste. It also may include hazardous waste such as asbestos, household chemicals and paints, automotive fluids, and commercial or industrial waste. The potential for contaminated runoff and ground water depend on such factors as the proximity of the dump to surface water, elevation of the ground water table, and permeability of the soil.

Other sources of contamination on public land include shooting ranges, formerly used defense sites, and wood treatment sites. Shooting ranges pose the potential for lead contaminates entering surface water and ground water. Acidic rainfall or acidic soil can dissolve the weathered lead compounds. In a dissolved state, lead can move through the soil and enter surface water and ground water. Shooting ranges in areas with acidic soils or acidic rainfall have an increased potential for transporting contaminates offsite and into drinking water. Bare ground on ranges may further increase the risk of migration of lead compounds offsite.

Sites once used by the Department of Defense (DOD) for military training and industrial facilities are on both public and private land. The DOD estimates that over 9,000 such sites exist. They pose a wide range of environmental hazards, including unexploded ordinances from the training sites and soil contamination from solvents, fuels, and other petroleum compounds used at industrial facilities. Sites are being cleaned up to minimize environmental effects. The Forest Service, for example, has identified over 100 formerly used defense sites on national forests.

Field treatment of wood posts is another past land-use activity that may have led to surface and ground water contamination. The common practice was to dip wooden posts into tanks that contained creosote, pentachlorophenol, or a chromium, copper, and arsenic compound and move them to an area for dripping and drying. The practice has been discontinued on public land such as national forests. However, a potential exists for surface and ground water contamination from past wood treatment operations.

Reliability and Limitation of Findings

Most available information on types of hazardous material activities and the contaminants associated with these activities is reliable because it is based on extensive site-specific data from Federal agency hazardous waste site cleanup programs. A limitation is that inventories identifying all hazardous waste sites on or adjacent to public land are incomplete. During field visits, areas of stressed vegetation, discolored or stained soil and water, indications of former structures or operations, and land disturbances may indicate the presence of old, abandoned, or illegal waste disposal sites. Due to the potentially hazardous nature of these disposal sites, discovery of such conditions should be reported to the appropriate agency official for further action (U.S. Department of Agriculture, Forest Service 1999). Other potential sources of ground water and surface water contamination, which should be considered in conducting source water assessments, are cemeteries and small airports and airstrips, especially those used for aerial application of chemicals.

Because the need for landfills exists, the design and management of safe landfills are paramount. To meet this need, Federal, State, tribal, and local governments have adapted an integrative approach that involves three waste management techniques: (1) decreasing the amount of waste through source reduction, (2) recycling of materials, and (3) improving design and management of landfills (U.S. EPA 1993). A number of regulations exist for the management of a municipal solid waste landfill, and many regulations have flexibility to meet local conditions; managers are advised to contact a local EPA or State agency office for information on siting, designing, and managing for their landfill.

Research Needs

Cleanup of ground water and soil contaminated by solid and hazardous wastes can be expensive and take long periods of time. Research is needed to develop more effective and cheaper cleanup methods.

Key Points

Several factors need to be considered when resource managers address the effects of landfills on drinking water quality:

1. Identification of landfill sites—proximity to wells, aquifers, geological and hydrological features, and surface waters.
2. Knowledge of the landfill age (a) old landfills—landfills existing before RCRA may contain hazardous material and may be improperly designed for hazardous material storage and municipal waste; (b) existing landfills—landfills existing after RCRA may still pose a problem for ground water contamination because the site may contain older units where hazardous waste was deposited improperly (these sites may have been improperly designed or may have punctured liners or clay layers); and (c) new landfills—landfills being managed under current Federal and State regulation should pose fewer problems, but small landfills may be exempted from certain regulations.
3. Knowledge of landfill history—What was deposited on the site and when? How was the landfill constructed? Does it have a clay layer, a liner, or a combination of the two?
4. Monitoring data—Is the site being monitored for VOC's and ground water contamination? Is monitoring sufficient to safeguard ground water sources?
5. Extent of contamination plume—If ground water is contaminated, what is the vertical and horizontal extent of the contamination? What is the effect of the plume on drinking water sources?
6. Compliance with current Federal and State regulations—What mitigation actions have been taken to comply with Federal and State laws if contamination occurred?

Literature Cited

Avci, C.B. 1992. Flow occurrence between confined aquifers through improperly plugged boreholes. Journal of Hydrology. 139 (1): 97–114.

Baker, J.A. 1998. What's in your groundwater? Waste Age. 29: 213–224.

Birch, P.B. 1995. New development standards in the Puget Sound Basin. In: National conference on urban runoff management: enhancing urban watershed management at the local, county, and State levels. Cincinnati, OH: U.S. Environmental Protection Agency, Office of Research and Development, Center for Environmental Research Information: 88–95.

Blomquist, P.K. 1984. Abandoned water wells in southern Minnesota. In: Proceedings of the 7th national ground water quality symposium: Innovative means of dealing with potential sources of groundwater contamination; 1984 September 26–28; Las Vegas, NV. Worthington, OH: National Water Well Association: 330–342.

Borden, R.C.; Yanoschak, T.M. 1990. Ground and surface water quality impacts of North Carolina sanitary landfills. Water Resources Bulletin. 26: 269–277.

Brown, K.; Donnelly, K.C. 1988. Estimation of the risk associated with the organic constituents of hazardous and municipal landfill waste. Hazardous Waste and Hazardous Materials. 5: 1–30.

Craun, G.F.; Calderon, R. 1996. Microbial risks in groundwater systems, epidemiology of waterborne outbreaks. In: Under the microscope: Proceedings of the groundwater foundations's 12th annual fall symposium; 1996 September 5–6; Boston. Denver: American Water Works Association Research Foundation: 9–15.

Gass, T.E. 1988. Impact of abandoned wells. Water Well Journal. 42 (1): 48–49.

Gleick, P.H. 1999. The World's water 1998–1999. Washington, DC: Island Press. 307 p.

Kaplan, B.O. 1991. Degradation of groundwater by septic tank systems. In**:** Kaplan, B.O., ed. Septic systems handbook**.** Chelsea, MI: Lewis Publishers: 133–144.

Kelley, M.; Phillips, N. 1995. Ordinances for the protection of surface water bodies: septic systems, docks, and other structures, wildlife corridors, sensitive aquatic habitats, vegetative buffer zones, and bank/shoreline stabilization. In: National conference on urban runoff management: enhancing urban watershed management at the local, county, and State levels; 1993 March 30–April 2; Chicago. Cincinnati: U.S. Environmental Protection Agency, Center for Environmental Research Information, Office of Research and Development: 96–109.

Livingston, E.H. 1993. Watershed planning and program integration. In: National conference on urban runoff management: enhancing urban watershed management at the local, county, and State levels; 1993 March 30–April 2; Chicago. Cincinnati: U.S. Environmental Protection Agency, Center for Environmental Research Information, Office of Research and Development: 1–12.

Nye, J.D. 1987. Abandoned wells: how one State deals with them. Water Well Journal. 41(9): 42–46.

Ogden, A.E.; Redman, R.K.; Brown, T.L. 1991. Preliminary assessment of the impact of class V injection wells on karst groundwater. In: Appalachian karst: Proceedings of the Appalachian karst symposium; 1991 March; Radford, VA. Huntsville, AL: National Speleological Society, Inc.: 211–216.

Phillips, N.J.; Lewis, E.T. 1995. Site planning from a watershed perspective. In: National conference on urban runoff management: enhancing urban watershed management at the local, county, and State levels; 1993 March 30–April 2; Chicago. Cincinnati: U.S. Environmental Protection Agency, Office of Research and Development, Center for Environmental Research Information: 139–150.

Rickabaugh, J.F.; Kinman, R.N. 1993. Evaluation of trace VOC emissions from sanitary landfills. Journal of Environmental Engineering. 119: 645–657.

U.S. Census Bureau. 1997. Estimates of the population of counties: annual time series, July 1, 1990, to July 1, 1996 (CO–96–1)(includes revised April 1, 1990, census population counts). Washington, DC: U.S. Census Bureau, Population Division, Statistical Information Staff. [Not paged]. [Available on the Web at: http://www.census.gov/population/censusdata/table-16.pdf and http://www.census.gov/population/projections/state/yr95to00/].

U.S. Department of Agriculture, Forest Service. 1999. Forest Service guide to land transactions. EM–2160–2. Washington, DC: U.S. Department of Agriculture, Forest Service, Engineering Staff. 27 p.

U.S. EPA. 1983. Results of the nationwide urban runoff program. PB84–185552. Washington, DC: U.S. Environmental Protection Agency. 188 p.

U.S. EPA. 1994. Drinking water treatment for small communities. EPA/V640/K–94/003. Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development. 48 p.

U.S. EPA. 1996. UST program facts. EPA–510–B–96–007. Washington, DC: U.S. Environmental Protection Agency. 18 p.

U.S. EPA. 1997a. Large-capacity septic systems. Draft document EPA–816–R–97–002. Washington, DC: U.S. Environmental Protection Agency, Office of Water. 38 p.

U.S. EPA. 1997b. Research plan for microbial pathogens and disinfection by-products in drinking water. EPA/600/–97/122. Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development, Office of Water: 3–1 to 3–52.

U.S. EPA. 1998a. Class V injection wells. EPA/816–F–98–009. Washington, DC: U.S. Environmental Protection Agency, Office of Water. 4 p.

U.S. EPA. 1998b. Overview of the Federal UST program. Washington, DC: U.S. Environmental Protection Agency. [Not paged]. [Available on the Web at: http://www.epa.gov/swerust1/overview.htm].

U.S. EPA. 1998c. Oxygenates in water: critical information and research needs. EPA/600/R–98/048. Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development. 65 p.

U.S. EPA. 1998d. The quality of our Nation's water: 1996. EPA–841–S–97–001. Washington, DC: U.S. Environmental Protection Agency. 197 p.

U.S. EPA. 1998e. Revisions to the underground injection control regulations for class V injection wells. Federal Register. 63(145): 40,586–40,619.

U.S. EPA. 1999. Ground water rule proposal. Draft document EPA 99–022. Washington, DC: U.S. Environmental Protection Agency. 78 p.

U.S. Geological Survey. 1999. The quality of our Nation's water—nutrients and pesticides: Circ. 1225. Denver: U.S. Geological Survey. 82 p.

Vitousek, P.M. 1994. Beyond global warming: ecology and global change. Ecology. 75: 1861–1876.

Weiss, K. 1995. Stormwater and the Clean Water Act: municipal separate storm sewers in the moratorium. In: National conference on urban runoff management: enhancing urban watershed management at the local, county, and State levels; 1993 March 30–April 2; Chicago. Cincinnati: U.S. Environmental Protection Agency, Center for Environmental Research Information, Office of Research and Development: 47–62.

Woldt, W.E.; Hagemeister, M.E.; Jones, D.D. 1998. Characterization of an unregulated landfill using surface-based geophysics and geostatistics. Ground Water. 36: 966–974.

Yates, M.V.; Yates, S.R. 1989. Septic tank setback distances: a way to minimize virus contamination of drinking water. Ground Water. 27: 202–208.

Yoder, C.O. 1995. Incorporating ecological concepts and biological criteria in the assessment and management of urban nonpoint source pollution. In: National conference on urban runoff management: enhancing urban watershed management at the local, county, and State levels; 1993 March 30–April 2; Chicago. Cincinnati: U.S. Environmental Protection Agency, Center for Environmental Research Information, Office of Research and Development: 183–197.

Zogorski, J.S.; Delzer, G.C.; Bender, D.A. [and others]. 1998. MTBE: summary of findings and research by the U.S. Geological Survey. In: Proceedings of the 1998 annual conference of the American Water Works Association—water quality; 1998 June 21–25; Dallas. Denver: American Water Works Association: 287–309.

Chapter 7

Concentrated Recreation

Myriam Ibarra and Wayne C. Zipperer[1]

Introduction

This chapter specifically examines drinking water issues related to concentrated recreation. The effects of concentrated recreation on drinking water quality encompass many topics with limited published literature. Because of limited space, selected components—campgrounds, ski resorts, water recreation, and traffic—are discussed in this report. However, land managers need to realize that these selected topics do not include all effects caused by concentrated recreation on drinking water and serve only to illustrate potential effects.

One of the most important attractions for public recreation is public land with natural cover. Increased demands for outdoor recreation result in greater needs for drinking water and in increased amounts of wastewater. Expanding recreation resorts invites larger numbers of visitors, and private tracts adjacent to public land are magnets for real estate development. This development may negatively affect drinking water and alter hydrologic processes. To illustrate the effects of concentrated recreation on drinking water supplies, we use data from the Forest Service, but our findings and recommendations are applicable to managers of other public and private land.

The National Forest System is the single largest supplier of public outdoor recreation in the United States. The national forests offer visitors 4,385 miles of national wild and scenic rivers; one-third of the National Wilderness Area System; about 8,000 miles of scenic byways; 133,000 miles of trails; more than 18,000 campgrounds, picnic areas, and visitor facilities; and 2.3 million acres of fishing lakes, ponds, and reservoirs. The Forest Service manages over 23,000 developed facilities, including campgrounds, trailheads, boat ramps, picnic areas, and visitor centers, in addition to permitted, privately owned facilities. These facilities can accommodate approximately 2.1 million people at one time. In 1997, the Forest Service hosted more than 800 million recreational visits that included skiing, hiking, camping, boating, fishing, hunting, and pleasure driving. The number is expected to grow to 1.2 billion by 2050.

The Forest Service manages over 3,000 drinking water systems. These systems range in complexity from hand pump wells to full water treatment plants at major installations. Primarily, these systems use ground water to provide drinking water at recreation sites and facilities. The Forest Service manages all public water systems in accordance with EPA and respective State regulations. In many cases, this approach exceeds minimum requirements for system operation.

The principal sources of pollutants produced by concentrated recreation are: (1) fuel residues from automobiles, watercraft, snowmobiles, and snow making machines; (2) wastewater from service facilities such as toilets, showers, restaurants, laundries, etc; and (3) soil and construction materials carried to surface waters with runoff at the time of construction. Detrimental effects of concentrated recreation are likely to be episodic or seasonal. The negative impacts of increased vehicular traffic and concentrated water recreation may be more apparent on surface water supplies, while the greater impact of concentrated winter recreation may be in ground water. This chapter deals with the effects of increased vehicular traffic, water recreation, and winter recreation.

Campgrounds

Issues and Risks

The effects of concentrated camping on drinking water quality are similar to those reported in chapter 8 for dispersed recreation. However, the magnitude, severity, and frequency of disturbance are greater with concentrated camping and the associated showers and toilets than with dispersed camping because of the greater density of humans using the site. Like other developments, the effects of a campground on drinking water quality depend on soil conditions, the presence of vegetation, and existing infrastructure.

[1] Hydrologist, 6655 Canton Street, Warner, NY; and Research Forester, USDA Forest Service, Northeastern Research Station, Syracuse, NY, respectively.

Findings

With the intense use of a site for camping, soil conditions become extremely important. Soils may lose their organic layer, become compacted, and become more erodable. Consequently, more surface erosion may occur as runoff increases. Without treatment to mitigate effects, the increase in erosion may result in increased stream turbidity and sedimentation. Techniques used to minimize soil compaction in urban parks (Craul 1992) may be applicable to campgrounds. Concentrated camping also could lead to streambank destabilization and further erosion and sedimentation. The proximity of campgrounds and picnic areas to water increases the chance of streambank erosion and destabilization as people use the water for swimming, bathing, and cleaning cooking and eating utensils.

Vegetation plays a key role in minimizing site degradation. Vegetation reduces erosion by slowing the movement of water across the ground surface and increases infiltration of water by decreasing soil compaction. However, with increased recreational use, vegetation presence decreases if active management does not occur to promote vegetation growth and reduce soil compaction (Craul 1992).

Unlike dispersed camping, concentrated campgrounds require infrastructure, including parking areas, restrooms, and shower facilities. This infrastructure may contribute to the contamination of surface water and ground water. Proper planning, design, and maintenance of facilities can minimize contamination of drinking water sources.

Contaminants associated with campers include fecal material, household cleansers and detergents, garbage and other floatables, cooking grease and oil, and antifreeze and motor oil. Because of their remote locations, campgrounds may serve as sites for illegal dumping of hazardous materials. Enforcement of clean water policies and educational programs may reduce the levels of these contaminants.

Water Recreation

Issues and Risks

Concentrated recreation on surface water produces chemical and microbial contamination. Individual boats, marinas, and swimmers usually release only small amounts of pollutants that can go undetected. When the number of participants is large, however, these sources can cause tangible water-quality problems in lakes, reservoirs, and rivers. Boating and marinas are associated with increased chemical pollutant concentrations and high levels of pathogens in the water (Gelt 1995, 1998). The effects of swimmers on drinking water supplies are an emerging problem that has prompted some utilities to limit or ban recreation on reservoirs used as drinking water sources. People with weak immune systems are particularly at risk because current methods for drinking water treatment do not detect or eliminate all pathogens, and some residues of chlorination are toxic.

Findings

The use of gasoline with methyl tertiary butyl ester (MTBE) in motorboats, particularly those using older two-cycle engines, contaminates surface water (U.S. EPA 1998a). An estimated 345 million motor boating trips and 29 million jet skiing trips occurred in the United States during 1994–95 (Cordell and others 1997). Nearly all personal watercraft and outboard motors use two-cycle engines. The fuel-inefficient design of two-cycle outboard motors is essentially unchanged since the 1930’s. Up to 30 percent of the gas used in the motor goes into the water unburned. Similarly, 10 percent of the fuel used by a personal watercraft, such as a Jet Ski, leaks into the water.

To assess the impact of two-cycle motorboat engines on water quality and aquatic life, scientists measured fuel residues in water in the Lake Tahoe Basin. They found MTBE; benzene, toluene, ethylbenzene, and xylene (BTEX); and polycyclic aromatic hydrocarbons (PAH’s) near shore in lakes that allow motorized watercraft. In open water, the concentrations of MTBE and BTEX were at or under the analytical detection limit. On sites with 50 to 100 watercraft engines, MTBE and benzene exceeded drinking water standards, but concentrations did not approach the criteria for protection of aquatic life. Concentrations decreased by the end of the boating season (Allen and others 1998).

Inefficient two-stroke carburetor engines used in personal watercraft and as outboard motors are the main source of fuel pollutants. These engines emitted more than 90 percent of the MTBE, 70 percent of benzene, and 80 percent of toluene into Lake Tahoe. In contrast, four-stroke inboard fuel-injected engines emitted an estimated 8 percent of MTBE, 28 percent of benzene, and 17 percent of toluene. Estimated volume of constituent load for Lake Tahoe during the 1998 boating season from two-stroke engines was in the order of thousands of gallons of MTBE, hundreds of gallons of benzene, and tens of hundreds of gallons of tolulene. There was no evidence that MTBE or BTEX were transported to the bottom of the lake or accumulated there (Allen and others 1998). Laboratory testing of newer engine technology suggested that emissions from marine outboard engines could be virtually eliminated by using more efficient Ficht injected engines (Allen and others 1998).

Proposed legislation in California moves the implementation date for stricter EPA emissions controls on personal watercraft engines up 5 years to 2001 from 2006.

Because marinas are located at the water's edge, pollutants can go directly to waterways. Water pollution from boating and marinas is linked to several sources. They include leaks from underground storage tanks, watercraft engines, and boat maintenance garages; discharge of sewage from boats; and stormwater runoff from parking lots (U.S. EPA 1993). Moreover, physical alteration of shorelines, wetlands, and aquatic habitat during the construction and operation of marinas may change flow patterns and result in poorly flushed waterways.

During boat maintenance, significant amounts of solvent, paint, oil, and other pollutants potentially can seep into ground water or be washed directly into surface water. Paints used to protect boats generally contain toxins that limit aquatic organism growth. Many boat cleaners contain chlorine, ammonia, and phosphates that harm plankton and fish. Small oil spills released from motors and refueling activities contain petroleum hydrocarbons that may attach to sediments. Hydrocarbons persist in aquatic ecosystems and harm bottom-dwelling organisms that are at the base of the aquatic food web. The EPA recommends that boaters use nontoxic cleaning products to reduce pollution. Boat owners can prevent pollution from paint and other chemicals by vacuuming up loose paint chips and paint dust and by using a drop cloth when cleaning and maintaining boats away from the water. Carefully fueling boat engines, recycling used oil, and discarding worn motor parts into proper receptacles can prevent needless petroleum spills. Most importantly, good engine maintenance prevents fuel and lubricant leaks and improves fuel efficiency (U.S. EPA 1993). Pollution from boating can potentially impair drinking water reservoirs or seep into ground water wells that provide drinking water along the shoreline.

Discharge of sewage and waste from boats can degrade water quality, especially in marinas with high boat use. Improper disposal of human and pet waste may introduce pathogenic bacteria, protozoans, and viruses into water (Gelt 1995, U.S. EPA 1993). Sewage from boats can make water unsuitable for recreation, destroy shell fishing areas, and cause severe human health problems. Sewage discharged from boats also stimulates algal growth, which can reduce the available oxygen needed by fish and other organisms. Although fish parts are biodegradable, large amounts of fish-cleaning remains can reduce water quality. Marinas should have adequate wastewater-disposal hook-ups and disposal sites for solid waste from boats. Well kept toilet facilities, designated pet areas, and health education postings also promote public health.

The locating and design of marinas are two of the most significant factors impacting water quality. Mastran and others (1994) found that inlets had higher concentrations of pollutants than the main channel, suggesting that hydrology plays a role in the distribution of the pollutants. Poorly placed marinas disrupt natural water flushing and cause shoreline soil erosion, habitat destruction, and consequently, degradation of water quality. Marinas should be located and designed to be regularly flushed by natural currents. Good design of a marina can provide an optimum combination of capacity, services, and access, while minimizing environmental impacts and onsite development costs (U.S. EPA 1993).

Concentrated swimming may cause microbial contamination of drinking water sources. A study conducted for the metropolitan water district of southern California determined that a swimmer or bather releases 0.1 gram of feces when entering the water; infants can add significantly more. Human feces may harbor viruses, bacteria, protozoa, and worm pathogens, some of which have been found in water treated by standard water purification methods. Bacteria are generally removed by present water treatments. Some viruses, like hepatitis A and Norwalk, are hardier and can be controlled only with additional amounts of disinfectant. See chapter 2 for further discussion on waterborne pathogens.

Water that is accidentally drunk while wading or swimming poses serious risks. Even small numbers of microbes may cause disease. It is estimated that in one outing a swimmer or wader ingests from 0.3 to 1.7 ounces of water that may be contaminated with feces (Gelt 1998). Outbreaks of Cryptosporidiosis have been documented from lakes, community and hotel pools, a large recreational water park, a wave pool, and a water slide. From January 1995 to December 1996, 37 outbreaks in 17 States were attributed to recreational water exposure. Diseases caused by *Escherichia coli* O157:H7*,* a specific strain of *E. coli* that is known to cause death if ingested, were associated primarily with recreational lake water. *Cryptosporidium* spp. and *Giardia* spp. were associated with a few outbreaks in swimming pools. Outbreaks of *Cryptosporidium* affected almost 10,000 people, and occurred in swimming pools that were chlorinated (Levy and others 1998).

It is difficult to estimate how many people become sick after contact with fecal contaminated water. For most people the symptoms are not acute. A person experiencing diarrhea, fever, vomiting, and nausea for 2 or 3 days may assume that he or she has the flu or ate some unsuitable food. In fact, a person may have a gastrointestinal infection acquired from drinking water (Gelt 1998). Epidemic outbreaks of waterborne disease have been recognized only after thousands of acute cases were reported (Levy and others 1998). Isolated and chronic waterborne diseases probably go undetected or unrecognized (also see chapter 6 on wastewater treatment).

Methods used to detect enteric pathogens are not always sensitive to low concentrations but very small numbers of microbes can cause illness (Gelt 1998). Routine microbiological testing may miss transient contamination by swimmers. Measures that can be taken to minimize fecal contamination include: (1) providing changing tables for infants in locker rooms, (2) providing adequate toilet and hand washing facilities, (3) posting signs against drinking water or defecating in the water, and (4) recommending against swimming for children with gastrointestinal illness. Unfortunately, other mammals defecating in a waterbody may introduce enteric pathogens (see chapters 14, 15). Hence, fecal contamination cannot be completely eliminated.

Winter Recreation

Issues and Risks

The increasing public demand for winter sporting opportunities has led to creation and rapid expansion of skiing resorts in forested watersheds (Brooke 1999). These facilities may alter the water quality of pristine environments. The National Ski Area Association estimates that 60 percent of all downhill skiing in the United States occurs on national forests. In cooperation with the 135 ski area operators, through the National Winter Sports Program, the national forests provided downhill skiing opportunities to approximately 31 million people in fiscal year 1997. The ski industry hopes to extend the ski season or even have the ski resorts open year-round (Hoffman 1998). Some ski resorts are proposing to develop facilities for summer outdoor recreation activities such as golf, swimming, and tennis. With ski resort expansion, real estate development also expands. To maintain predictable revenues in spite of unpredictable weather, ski resorts increasingly rely on artificial snow to cover the slopes. While there is not an apparent direct effect of skiing on drinking water, environmentalists warn that large ski resorts alter natural hydrological cycles, increase traffic congestion, and are magnets for urban sprawl, all of which may impair water quality.

Findings

To satisfy public demand, the Forest Service is authorizing the development or expansion of ski resorts. For example, between January 1997 and January 1999, the EPA Office of Federal Activities filed environmental impact statements for work on 12 ski resorts inside of national forests. Development of ski resorts includes new construction or expansion of parking lots and service roads, downhill ski runs, cross-country ski trails, snowmobile trails, chair lifts, lodges, restrooms, ski patrol facilities, ski schools, ski repair shops, stores, hotels, and restaurants (U.S. Department of Agriculture, Forest Service 1992, 1998, 1999). The construction and operation of ski facilities affect drinking water sources in various degrees. Clearing of vegetation for ski runs increases the chances of soil erosion and hence higher turbidity and sedimentation in streams (Hoffman 1998). Pollutants from car emissions are deposited on the soil with precipitation. Runoff from roads, parking lots, or lawns may be contaminated with salt, heavy metals, petroleum residues, or landscaping chemicals. Expansion of impervious surfaces leads to increased peak runoff and shorter resident time of water in the watershed.

Newly developed ski resorts may cause shortages or dramatic fluctuations in drinking water supplies. Some resorts are projecting to host 5,000 to 10,000 visitors a day. The typical average consumption rate of water at ski areas is 10 gallons per day per skier capacity; if water conservation measures are in place, the intake could be reduced to 7 gallons per day. Thus, a ski resort with 13,000 skiers may need between 94,500 and 135,000 gallons per day (U.S. Department of Agriculture, Forest Service 1998). At the same time, a small but irretrievable loss of ground water may occur due to evaporation and sublimation from snow making (Hoffman 1998). To prevent artificially drastic pulses in downstream flow and to maintain channel stability, ski resorts may need to stop making snow when natural water levels are too low, or use water stored in ponds or lakes (U.S. Department of Agriculture, Forest Service 1997).

Ski resorts are often located on environmentally sensitive sites. In mountainous regions, the slopes are steep, the soils are thin, the subsurface is predominantly gravel and cobble, and the aquifers are fractured bedrock. This type of aquifer is very sensitive to pollution because the rapid ground water flow can carry microbes and other pollutants for long distances (U.S. EPA 1999). Ski resorts have a special problem with wastewater treatment. The peak need is in the winter, when conventional sewage treatment methods function at slower rates and microbial pathogens survive longer in water and soil. One solution is to build storage ponds and apply wastewater treatment in warmer weather. Such storage, however, is not always economically or logistically feasible. Another method being tested makes artificial snow from wastewater and stores the snow on slopes where skiing is not permitted (Gibson 1996). In ideal conditions, the wastewater stored would melt and percolate very slowly, producing a clean effluent. However, sudden snowmelt could contaminate surface water and ground water with effluent.

Ski resorts rely more and more on snow making and grooming to attract skiers. Some snow making operations require massive amounts of stream water. To get enough water, resorts have relocated stream channels, excavated wells, constructed ponds or pumped water from neighboring surface water sources. Each activity may alter the natural flow of water and ultimately influence drinking water quality. Not only is water being redistributed to another location, the generators that power the snow machines and pumps may contribute to air pollution. For instance, the diesel generators in one resort in Vermont are the eighth largest air polluters in the State (Hoffman 1998). This pollution may contribute to atmospheric deposition of contaminants.

Increased Traffic

Issues and Risks

Vehicular traffic in forests and grasslands creates fuel emissions that are deposited on the ground through wet and dry deposition. Pollution from fuel emissions may migrate to surface water and ground water through rain or snowfall. The most significant sources of fuel pollutants are cars, but in some places all-terrain vehicles and snowmobiles also are important contributors. In the last decade, the number of recreational visits to national forests increased by 40 percent, and the number of visits was highly correlated with the number of vehicles (Cordell and others 1997). Additionally, tourism to recreational resorts promotes urbanization, which in turn adds traffic. For example, in the Eisenhower Tunnel connecting Denver with the busiest ski areas in Colorado, the traffic has quadrupled in the last 25 years. Improvement and expansion of parking lots and roads increase peak runoff and nonpoint-source pollution from impervious surfaces. Runoff may be contaminated with salt, heavy metals, petroleum residues, or landscaping chemicals that can degrade surface water and ground water quality (U.S. EPA 1983). Oxygenates and PAH's are gasoline residues that have been found in drinking water supplies and are potential threats to human health. Deposition of MTBE, an oxygenate, may be especially significant during the winter because concentrations of MTBE in precipitation are higher at colder than warmer temperatures (Delzer and others 1996). Widespread impacts may result from vehicular emissions that dissolve in rain or snowfall and subsequently infiltrate to shallow ground water. Additional research needs to be conducted to determine the significance of increased auto emissions on drinking water quality in rural areas.

Auto emissions also contribute to the amount of nitrogen in the atmosphere. Nitrogen deposition from the atmosphere varies across the country with the greatest concentrations occurring in a broad band from the Upper Midwest through the Northeast (U.S. Geological Survey 1999). Recent studies have shown that atmospheric deposition of nitrogen can be quite significant. For example, approximately 25 percent of the nitrogen entering the Chesapeake Bay Estuary comes from the atmosphere (Fisher and Oppenheimer 1991). The effect of nitrogen deposition on drinking water is an area that needs further research (see chapter 3). See chapter 6 for further discussion of urban runoff and MTBE.

Reliability and Limitation of Findings

The potential negative impacts of concentrated recreation on drinking water supplies have been recognized and addressed in a qualitative way, but quantitative assessments are very rare. The material presented here comes almost exclusively from government reports and newspaper articles rather than from the primary scientific literature. This fact suggests that the issue has not been subjected to adequate scientific investigation.

A simplistic first approximation is to consider the expansion of concentrated recreation in forests as small-scale urbanization. However, it is important to keep in mind that the toxicities of some pollutants produced by recreational activities have been measured only in the laboratory. Furthermore, survey data on impacts on water quality by recreation are mostly from water that is not used for drinking (Cox 1986, Gelt 1995). The extremely varied ecology of each forest together with the diverse nature of recreation activities suggests specific analysis for each situation. Drinking water sources seldomly appear to be susceptible to long-term degradation because of recreation, but some lakes and well water probably are susceptible to episodes of local pollution (Peavy and Matney 1977). Environmental impact statements are prepared when designing recreation resorts, and they often present plans to monitor surface water and ground water. In the absence of specific studies, analysis of these data could be the first step in describing regional or national patterns.

The EPA and Centers for Disease Control and Prevention (CDC) recognized that waterborne diseases are common in the United States (Levy and others 1998), but data on their occurrence are very sparse. The United States has recorded incidences of waterborne diseases only since 1985. The EPA and CDC are conducting a series of pilot studies to produce the first national estimate of waterborne disease occurrence (U.S. EPA 1998b). Of particular importance are the levels of disease associated with drinking water that otherwise meet Federal and State standards. This research would also serve as a springboard for more localized assessments of drinking water quality.

General principles of urban water pollution are applicable in expanding recreation resorts in all regions. However, data to quantify impacts on specific sites are not readily available. The Forest Service and others have monitoring programs that document some aspects of water resources, but we are not aware of any efforts to collect data specifically to evaluate the impact of concentrated recreation on drinking water supplies.

Research Needs

The field of recreation ecology is relatively new. Only recently have scientists begun to study the relationships among use-related, environmental, and managerial factors (Marion 1998). Evaluation of the effect of recreation on drinking water could be approached through monitoring the effects of the visitor population and the impacts of population growth in communities adjacent to recreation sites.

1. One basic task is to document the kinds of data that have been collected as part of routine water-quality monitoring and sanitary engineering operations.
2. The next step is to design a sampling program for evaluating impacts of recreation on drinking water supplies.
3. Impacts also need to be assessed across the range of scale from local to major watershed.

Key Points

1. Concentrated recreation, like urbanization, affects water quality through wastewater treatment and urban runoff.
2. Ski resorts alter hydrologic processes by changing the availability of water during the year.
3. Decreased streamflows may increase the concentration of contaminants from wastewater and runoff.
4. Wastewater treatment is especially precarious for ski resorts because peak treatment is during the winter.
5. Water recreation, both swimming and boating, may have direct effects on drinking water quality at the local scale.
6. Increased traffic may affect drinking water quality through deposition of MTBE and nitrogen.

Literature Cited

Allen, J.; Reuter, J.; Goldman, C. [and others]. 1998. Lake Tahoe motorized watercraft report—Integration of water quality, watercraft use and ecotoxicology issues. Davis, CA; University of California; Reno, NV: University of Nevada; preliminary draft report. 24 p.

Brooke, J. 1999. Environmentalists battle growth of ski resorts. The New York Times. January 19: Sec. A; 10 (col. A1).

Cordell, H.K.; Teasley, J.; Super, G. [and others]. 1997. Results from the national survey on recreation and the environment, all Forest Service regions. Athens, GA: U.S. Department of Agriculture, Forest Service; University of Georgia. [Available on the Web at: http:/www.fs.fed.us/research/rvur/recreation/].

Cox, E.R. 1986. Wastewater movement near four treatment and disposal sites in Yellowstone National Park, Wyoming. WRI–84–4356. Denver: U.S. Geological Survey. 81 p.

Craul, P.J. 1992. Urban soils in landscape design. New York: John Wiley. 396 p.

Craun, G.F.; Calderon, R. 1996. Microbial risks in groundwater systems, epidemiology of waterborne outbreaks. In: Under the microscope: Proceedings of the groundwater foundations's 12th annual fall symposium; 1996 September 5–6; Boston. Denver: American Water Works Association Research Foundation: 9–15.

Delzer, G.C.; Zogorski, J.S.; Lopes, T.J. [and others]. 1996. Occurrence of the gasoline oxygenate MTBE and BTEX compounds in urban stormwater in the United States, 1991–1995. Rep. 96–4145. Denver: U.S. Geological Survey, Water-Resources Investigations. 8 p.

Fisher, D.C.; Oppenheimer, M. 1991. Atmospheric nitrogen deposition and the Chesapeake Bay Estuary. AMBIO. 20(3–4): 102–108.

Gelt, J. 1995. Water recreation makes big splash in Arizona. Arroyo. 7: 1–11.

Gelt, J. 1998. Microbes increasingly viewed as water quality threat, the emerging contaminants. Arroyo. 10: 1–14.

Gibson, T. 1996. A cool solution for treating wastewater. Compressed Air. 101(8): 18–22.

Hoffman, C. 1998. Let it snow. Smithsonian. 29: 50–53.

Levy, D.A.; Bens, M.S.; Craun, G.F. [and others]. 1998. Surveillance for waterborne-disease outbreaks—United States, 1995–1996. Morbidity and Mortality Weekly Report. 47(SS–5): 1–34.

Marion, J.L. 1998. Recreation ecology research findings: implications for wilderness and park managers. In: Proceedings of the national outdoor ethics conference; 1998 April 18–21; St. Louis. Gaithersburg, MD: Izaac Walton League of America: 188–196.

Mastran, T.A.; Dietrich, A.M.; Gallagher, D.L. [and others]. 1994. Distribution of polyaromatic hydrocarbons in the water and sediments of a drinking water reservoir with respect to boating activity. Water Research. 28: 2353–2366.

Peavy, H.S.; Matney, C.E. 1977. The effects of recreation on water quality and treatability; drinking water quality enhancement through source protection. In: Pojasek, R.B., ed. Drinking water quality enhancement through source protection. Ann Arbor, MI: Ann Arbor Science Publishers Inc.: 461–475.

U.S. Department of Agriculture, Forest Service. 1992. Loon Mountain ski area, South Mountain expansion project, Grafton County, NH. Final environmental impact statement. 247 p. On file with: White Mountain National Forest, Federal Building, 719 Main Street, Laconia, NH 03246–0772.

U.S. Department of Agriculture, Forest Service. 1997. Record of decision for the Waterville Valley Ski Resort development of snowmaking water impoundments and environmental impact statement. 39 p. On file with: White Mountain National Forest, Federal Building, 719 Main Street, Laconia, NH 03246–0772.

U.S. Department of Agriculture, Forest Service. 1998. Pelican Butte ski area master development plan: implementation. WIN–24–98: III–19 to III–83. On file with: Winema National Forest, 2819 Dahlia, Klamath Falls, OR 97601.

U.S. Department of Agriculture, Forest Service. 1999. Final environmental impact statement for permitting Appalachian Mountain Club (AMC) huts and Pinkham Notch Visitor Center (PNVC) in the White Mountain National Forest. 279 p. On file with: White Mountain National Forest, Federal Building, 719 Main Street, Laconia, NH 03246–0772.

U.S. EPA. 1983. Results of the nationwide urban runoff program. PB84–185552. Washington, DC: U.S. Environmental Protection Agency. 188 p.

U.S. EPA. 1993. Guidance specifying management measures for sources of nonpoint pollution in coastal waters. EPA–840–B–93–001c. Washington, DC: U.S. Environmental Protection Agency, Office of Water, Nonpoint Source Pollution Control Program. 828 p.

U.S. EPA. 1998a. Class V injection wells. EPA/816–F–98–009. Washington, DC: U.S. Environmental Protection Agency, Office of Water. 4 p.

U.S. EPA. 1998b. Overview of the Federal UST program. Washington, DC: U.S. Environmental Protection Agency. [Not paged]. [Available on the Web at: http://www.epa.gov/swerust1/overview.htm].

U.S. EPA. 1999. Ground water rule proposal. Draft document EPA 99–022. Washington, DC: U.S. Environmental Protection Agency. 78 p.

U.S. Geological Survey. 1999. The quality of our Nation's water—nutrients and pesticides: Circ. 1225. Denver: U.S. Geological Survey. 82 p.

Chapter 8

Dispersed Recreation

David Cole[1]

Introduction

Dispersed recreation is a common and growing use of forests and grasslands that has the potential for significant impacts on the quality of public drinking water sources.

Issues and Risks

Trails are constructed to provide access. Visitors walk, ride, and bicycle along trails. Runoff from trails can add sediment to streams, particularly at trail fords. Visitors picnic, camp, and walk or ride off-trail; in some places, they use off-road vehicles to travel cross country. The resultant loss of vegetation and compaction of soil can lead to increased runoff, erosion, and sedimentation. Visitors who pull off roads to view scenery, picnic, camp, or access the immediate surroundings can cause increased erosion and sedimentation of streams. Where people pull off roads to picnic or camp adjacent to streams, foot traffic and vegetation loss on streambanks can result in streambank erosion and channel instability.

Visitors and their animals can contaminate water supplies by carrying and depositing feces containing microorganisms that cause human diseases. Contamination comes from fecal deposition and from direct contact with water during activities, such as swimming and washing. Recreational behaviors are commonly unrestricted, visitor education is typically inadequate, and where activities are dispersed, few facilities are provided to ensure proper disposal of human waste. Consequently, drinking water quality problems associated with recreation use may be expected. In a recent survey of Forest Service watershed managers, recreation was the most commonly reported cause of water-quality concerns. However, this high frequency of concern does not necessarily mean that recreation is the most common or serious source of water contamination in national forest watersheds.

[1] Research Biologist, Aldo Leopold Wilderness Research Institute, Missoula, MT.

Findings from Studies

The impacts of dispersed recreation on sediment have not been systematically quantified. Recreation facilities (particularly trails) and recreation use elevate sediment levels (see chapter 9). Nonmotorized recreation simply does not disturb much of the watershed. Cole (1981) found, for example, that <0.5 percent of a heavily used portion of the Eagle Cap Wilderness in Oregon was directly affected by trails and camping. Most of the disturbed area was located far enough from streams so that the effect was negligible. Recent research indicates that sediment yield from trails is much higher when trails are used by horses than by hikers or llamas (DeLuca and others 1998).

Impacts of dispersed motorized recreation activities on sediment, while not well quantified, are more likely to be significant. Impacts will vary greatly with such factors as type of vehicle, driving behavior, topography, vegetation type, soil erodibility, and climate. Both the extensiveness and the intensiveness of impact are much greater with motorized recreation than with nonmotorized recreation. In the extreme case of an off-road vehicle area in California, erosion rates were estimated to be 52 tons per acre per year (116.5 metric tonnes per hectare per year) (Wilshire and others 1978).

Pathogenic organisms can be introduced by recreationists into watersheds in which dispersed recreation is the primary land use. In a broad survey of surface municipal drinking water sources, LeChevallier and others (1991) found oocysts of *Cryptosporidium* spp. and cysts of *Giardia* spp. species even in protected watersheds. Suk and others (1987) found cysts of *Giardia* in 27 of 78 samples from back-country streams in several large wilderness areas in the Sierra Nevada in California. Taylor and others (1983) found *Campylobacter jejuni* in the stools of 23 percent of people reporting diarrhea and *G. lamblia* in the stools of 8 percent of such people. They also found these organisms in streams in the Grand Teton National Park, WY.

It is generally accepted, although still controversial, that mammals other than humans can spread these pathogenic organisms to humans. Since horses, mules, and dogs are more likely than humans to defecate directly in or near water, they may be a major concern if they are important disease carriers. Taylor and others (1983) found *Campylobacteria* in a sample of packstock stool in Grand Teton National Park, as well as in samples from humans. More than 40 mammals, both wild and domestic, have been found to harbor *Cryptosporidium parvum* (Current 1987). This evidence, along with the finding that *C. parvum* readily crosses host species barriers, has convinced most experts that human infections are often the result of transmission from wild and domestic animals, including horses and dogs (Current and Garcia 1991, Rose 1990). As for *Giardia*, Hibler and Hancock (1990) state, "some investigators considered the parasite found in humans (*Giardia lamblia)* to be host-specific, but the majority of the research performed to date questions this assumption." *Cryptosporidium* and *Giardia* have been found in humans and a wide variety of birds, mammals, fish, and reptiles (see appendix D). It has been cross transmitted between humans and a number of these animals (Hibler and Hancock 1990).

The issue of transmission by wild animals (chapter 14) is also relevant to the question of whether or not water quality can be adequately protected by eliminating or severely restricting recreation use. Questions remain about which pathogens are transmitted and the relative importance of humans and other animals as agents of transmission. Consequently, management actions such as the improvement of human waste disposal behavior and facilities, and even outright elimination of recreation use, while likely to reduce the transmission of disease organisms, are unlikely to eliminate the problem.

Studies that have attempted to relate intensity of recreation use to degree of water contamination have produced mixed results. Some studies report positive correlations (e.g., Suk and others 1987), others report no correlation (e.g., Silverman and Erman 1979), and at least one series of studies reports a negative correlation (Stuart and others 1971, Walter and Bottman 1967). One potential explanation for these divergent findings is that wild animal contamination may dwarf the effects of low levels of recreation. Indeed, some authors have noted that as levels of contamination increase, the strength of positive correlations between recreational use and contamination and between fecal coliform and the occurrence of *Giardia* and *Cryptosporidium* also increase (LeChevallier and others 1991).

The study finding a negative correlation between recreation use and bacterial contamination of water supplies initially compared a watershed closed to recreation use with a watershed open to use. Fecal coliform and fecal streptococci counts were higher in the closed watershed (Walter and Bottman 1967). After the watershed was opened to recreation and limited logging, bacterial contamination decreased. They concluded, "…these human activities drove from the watershed a large wild animal population which had contributed substantially to the previous bacterial population" (Stuart and others 1971: 1048).

From these findings, several implications can be drawn. First, surface water is not likely to be safe for drinking without purifying treatment, even where recreation use is excluded. In fact, Suk and others (1987) found in wilderness watersheds that 45 percent of high-use samples contained *Giardia* cysts, and 17 percent of the low-use samples contained cysts. Back-country visitors are advised to purify drinking water obtained from all surface water sources, regardless of the level of recreation use in the vicinity (Cilimburg and Monz, in press). Adequate purifying treatment for public drinking water may be expensive.

Second, it is more critical to improve management of recreation use and of human waste disposal in heavily used than in lightly used watersheds. Management options for areas with heavy dispersed recreation use include reducing recreation use, prohibiting pack animals and pets, providing adequate toilet facilities, and educating visitors in appropriate waste disposal techniques (see, e.g., Hampton and Cole 1995, Meyer 1994).

The relationship between the amount of dispersed recreation and water contamination depends on other variables including the type of recreation use, soils, slope, and climate. None of these relationships has been systematically evaluated. It is difficult to determine if recreation use is heavy or light, or to confidently prescribe management in field situations.

The importance of educating visitors in the proper disposal of human waste is suggested by studies of the survival of bacteria in feces buried in soil in Montana. Samples of feces were inoculated with two bacteria, *Escherichia coli* and *Salmonella typhimurium,* and both survived in large numbers for 8 weeks after burial in early summer (Temple and others 1980). Moreover, substantial numbers of *Salmonella* survived over winter. Depth of burial had no effect on persistence, and differences among burial sites were minor (Temple and others 1982). Clearly, the idea that shallow burial (in catholes) renders feces harmless in a short time is inaccurate. Removal of feces is the best means of disposal if toilets are not provided. The second best option is careful and complete burial far from water sources, campsites, and other heavily visited locations.

Reliability and Limitation of Findings

There is strong evidence to support the general findings that (1) dispersed recreation use can adversely affect the quality of surface drinking water supplies and (2) surface drinking water supplies will contain pathogenic microorganisms even in the absence of recreation use. Our ability to quantify the effect of dispersed recreation is very limited, as is our understanding of the importance of recreation as a source of contamination. Consequently, there is a weak foundation in science for decisions about where recreation use should be prohibited or restricted and where sanitary facilities should be provided or improved.

These general findings should be broadly applicable throughout the United States. Specifics of quantitative relationships between recreation use and water quality will vary with many environmental parameters. Logic suggests that one important regional distinction can be made between arid and mesic regions. In arid lands, visitors and animals are particularly drawn to water sources, increasing the likelihood of contamination and the effects may persist longer because these systems are not flushed rapidly or frequently.

Research Needs

1. We need to know if some pathogens, such as human enteric viruses, pose a significant threat to human health. As Gerba and Rose (1990) note, even though there are few cases where virus isolations in source water have been linked to human disease, there are many reasons to suppose that there is much more illness due to viral contamination than is recognized. We need a better understanding about the mechanisms of transmission for different pathogenic microorganisms, especially their presence in recreation pack animals, pets, and wild animals. Further research on *C. parvum* and *G. lamblia* is particularly important.
2. Additional research is needed to provide a more solid foundation for decisions about where and how to restrict dispersed recreation and where to invest in more and better sanitary facilities. We need better quantification of the relationship between drinking water microbiology and amount of use by visitors, their pets, and their pack animals. Thresholds of use need to be identified, above which adverse effects on water quality become pronounced and unacceptable. We need a better understanding of how site variables influence susceptibility to contamination and whether water-contact activities, such as swimming, are a significant concern at the low densities typical of dispersed recreation sites.
3. Research is needed to develop techniques capable of distinguishing between human and other sources of pathogens. Finally, we should assess (1) the validity of rules of thumb managers use to develop management prescriptions and (2) the effectiveness of techniques managers develop to mitigate contamination.
4. More research is needed on the decomposition rates of human feces, on variables that influence decomposition rates, and on how pathogens disperse in and over the soil. This information could contribute to better educational material about where and how to bury feces, and to better decisions about where sanitary facilities are needed.

Key Points

1. Since dispersed recreation can contribute to contamination, every affordable effort should be made to educate visitors in appropriate human waste disposal and to provide well-designed and appropriately located facilities for the disposal of human waste. Surface water from wildlands including wilderness, can contain pathogens that cause human disease unless drinking water is adequately treated. For public water supplies, adequate treatment may be expensive.
2. Where recreation use is high and water contamination is too high, sanitary facilities need to be developed or improved, and/or use of the area must be restricted. Where it is clear that dispersed recreation use is low, use restrictions and the provision of sanitary facilities are not worth the costs involved. In areas of moderate use, our understanding is inadequate to suggest whether it is worth the costs of limiting access, restricting behavior, or investing in sanitary facilities. Inadequate understanding also makes it difficult to identify use thresholds above which (1) rudimentary sanitary facilities are needed or (2) developed sanitary facilities are needed.

Literature Cited

Cilimburg, Amy; Monz, Christopher [In press]. Wildland recreation and human waste: a review of problems, practices and concerns. Environmental Management.

Cole, David N. 1981. Vegetational changes associated with recreational use and fire suppression in the Eagle Cap Wilderness, Oregon: some management implications. Biological Conservation. 20: 247–270.

Current, William L. 1987. *Cryptosporidium*: its biology and potential for environmental transmission. Critical Reviews Environmental Control. 17: 21–51.

Current, William L.; Garcia, Lynne S. 1991. Cryptosporidiosis. Clinical Microbiology Reviews. 4: 325–358.

DeLuca, T.H.; Patterson, W.A.; Freimund, W.A.; Cole, D.N. 1998. Influence of llamas, horses,and hikers on soil erosion from established recreation trails in western Montana, U.S.A. Environmental Management. 22: 255–262.

Gerba, Charles C.; Rose, Joan B. 1990. Viruses in source and drinking water. In: McFeters, Gordon A., ed. Drinking water microbiology. New York: Springer-Verlag: 80–396.

Hampton, Bruce; Cole, David. 1995. Soft paths. 2d ed. Mechanicsburg, PA: Stackpole Books. 222 p.

Hibler, Charles P.; Hancock, Carrie M. 1990. Waterborne giardiasis. In: McFeters, Gordon A., ed. Drinking water microbiology. New York: Springer-Verlag: 271–293.

LeChevallier, M.W.; Norton, W.D.; Lee, R.G. 1991. Occurrence of *Giardia* and *Cryptosporidium* spp. in surface water supplies. Applied Environmental Microbiology. 57: 2610–2616.

Meyer, Kathleen. 1994. How to shit in the woods. Berkeley, CA: Ten Speed Press. 77 p.

Rose, Joan B. 1990. Occurrence and control of *Cryptosporidium* in drinking water. In: McFeters, Gordon A., ed. Drinking water microbiology. New York: Springer-Verlag: 294–321.

Silverman, G.; Erman, D.C. 1979. Alpine lakes in Kings Canyon National Park, California: baseline conditions and possible effects of visitor use. Journal of Environmental Management. 8: 73–87.

Skinner, Q.D.; Adams, J.C.; Rechard, P.A.; Beetle, A.A. 1974. Effect of summer use of amountain watershed on bacterial quality. Journal of Environmental Management. 3: 329–335.

Stuart, D.G.; Bissonnette, G.K.; Goodrich, T.D.; Water, W.G. 1971. Effects of multiple use on water quality of high-mountain watersheds: bacteriological investigations of mountain streams. Applied Microbiology. 22: 1048–1054.

Suk, Thomas J.; Sorenson, Stephen K.; Dileanis, Peter D. 1987. The relation between human presence and occurrence of *Giardia* cysts in streams in the Sierra Nevada, California. Journal of Freshwater Ecology. 4: 71–75.

Taylor, D.N.; McDermott, K.T.; Little, J.R. [and others]. 1983. *Campylobacter* enteritis from untreated water in the Rocky Mountains. Annals of Internal Medicine. 99: 38–40.

Temple, Kenneth L.; Camper, Anne K.; Lucas, Robert C. 1982. Potential health hazard from human wastes in wilderness. Journal of Soil and Water Conservation. 37: 357–359.

Temple, Kenneth L.; Camper, Anne K.; McFeters, Gordon A. 1980. Survival of two enterobacteria in feces buried in soil under field conditions. Applied and Environmental Microbiology. 40: 794–797.

Walter, W.G.; Bottman, R.P. 1967. Microbiological and chemical studies of an open and closed watershed. Journal of Environmental Health. 30: 159–163.

Wilshire, H.G.; Nakata, J.K.; Shipley, S.; Prestegaard, K. 1978. Impacts of vehicles on natural terrain at seven sites in the San Francisco Bay area. Environmental Geology. 2: 295–319.

Chapter 9

Roads and Other Corridors

W.J. Elliot[1]

Introduction

The focus of this chapter is on the impact on drinking water quality of roads and other corridors, such as trails, utility rights-of-way, railroads, and airfields in forest and grassland watersheds. These corridors are essential for a wide range of access including residential, recreational, and managerial (U.S. Department of Agriculture, Forest Service 1998). They can be public and managed by Federal, State, or local agencies or private and managed by individuals or industries.

Roads, railroads, and similar corridors are major features in most watersheds. Figure 9.1 is a diagram of a typical insloping forest road in steep terrain. Water from the roadway is diverted to the ditch, and then directed to a culvert or surface drain. In less steep areas, or for larger roads, there are usually ditches on both sides of the road to collect and channel runoff. The runoff is then delivered to vegetated slopes for infiltration or to a natural channel that is part of the stream system (Packer and Christensen 1977).

Roads and similar corridors can adversely affect water quantity and quality in several ways. Runoff is low from undisturbed forests, but runoff rates from rainfall and snowmelt are greater from compacted road surfaces than from less disturbed parts of watersheds (Elliot and Hall 1997). The roadway, the ditch, and in some cases, the waterway below a road culvert are the main sources of detached sediment (fig. 9.1) from erosion depending on road surface material (Elliot and Tysdal 1999). The cutslopes and fillslopes erode mainly by mass wastage.

Eroded sediment is usually deposited on the undisturbed surface below the road (Elliot and Tysdal 1999, McNulty and others 1995, Packer and Christensen 1977). Establishing a buffer zone of undisturbed forest between a corridor and a stream is helpful, but if runoff from roads or other disturbances is channeled, or filter strips are too narrow, then buffer zones cannot be expected to eliminate sediment movement to streams. Most surface water contaminants enter streams at stream crossing by roads, railroads, or pipelines, or places where other disturbances are close to streams. Corridor-related disturbances also can degrade ground water from shallow wells, particularly in highly porous geologies, such as karst (Gilson and others 1994, Hubbard and Balfour 1993, Keith 1996).

Excavation at the bottom of a cutslope can intercept ground water, creating instability of the road or the cutslope and altering hydrology (Jones and others, in press). This intercepted ground water may also be affected by acid drainage (chapter 18). All of the excavated surfaces revegetate slowly and are prone to erosion (Burroughs and King 1989, Grace and others 1998).

When roads or other compacted corridors are abandoned, they can continue to be sources of sediment through chronic surface erosion or mass failure (Elliot and others 1996). Compaction of a disturbed surface frequently restricts vegetation regrowth. Bare surfaces are susceptible to erosion, and steep areas without trees are susceptible to landslides. In some cases, local frost heaving or minor slumping of fill shoulders can cause surface water to collect, leading to saturation of the fill and an increased risk of mass failure. Both surface erosion and mass failure can lead to

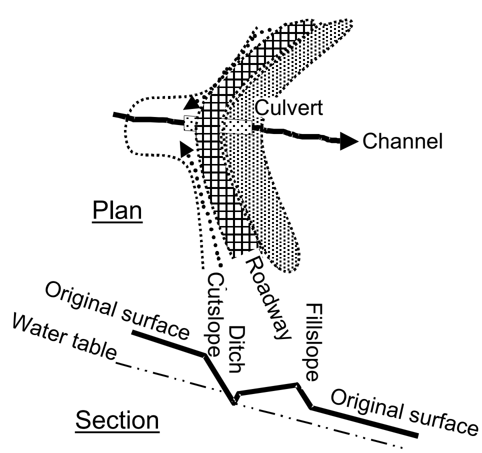


Figure 9.1—Surfaces and flow paths associated with a road cross section.

[1] Project Leader, USDA Forest Service, Rocky Mountain Research Station, Moscow, ID.

increased sediment loads in streams (McClelland and others 1998).

Runoff and seepage from roads and rights-of-way can contain elevated levels of sediment, metals, and complex hydrocarbons from the highway material and traffic. They may also contain traces of pesticides or other undesirable substances. Chemicals may be dissolved in the runoff water, but they frequently are attached to the eroded sediment particles.

Altered Hydrology

Issues and Risks

The presence of roads in a watershed may increase the frequency and magnitude of peak runoff discharges, particularly on small watersheds. Roads may also increase total runoff and decrease the time to peak runoff from major storms or snowmelt.

Findings from Studies

Roads have a number of impacts on hydrology. They intercept precipitation and snowmelt and, because they have lower infiltration rates, divert it as surface runoff to channels (Packer and Christensen 1977). Cutslopes (fig. 9.1) can also tap into ground water and divert it, increasing runoff. In a study of spring snowmelt in the northern Rockies, 58 percent of the runoff from a road was due to intercepted subsurface flow (Burroughs and Marsden 1972). Megahan (1983) found that road segments on granitic soils in central Idaho collected about 8.4 inches [21 centimeters (cm)] of water in subsurface flow from the area above the road. Road ditches can extend the stream network, increasing the volume of water available during the early part of a storm. The presence of roads can also shorten the time to peak flow during a runoff event (Wemple and others 1996). This diversion of ground water can dry out hillsides below the road, altering vegetation, and reducing water yield during dry periods later in the year.

If a road culvert is too small, or becomes blocked, water can be diverted from one subwatershed to another. Severe ditch and channel erosion may result (Megahan 1983; U.S. Department of Agriculture, Forest Service 1998). The cumulative effect is an increase in frequency and magnitude of peak discharges (Jones and Grant 1996; Megahan 1983; U.S. Department of Agriculture, Forest Service 1998).

Adding gravel to the road surface increases the porosity and roughness of the road, increasing the conductivity from under 1 millimeter (mm) per hour to 3 mm per hour or more (Foltz 1996). This results in decreased runoff rates from low-intensity rain and snowmelt. Gravel addition will have less impact during high-intensity storms. Ripping closed roadways can increase infiltration rates, but studies show that rates do not reach undisturbed levels (Luce 1997). Culverts or surface drainage structures to deliver water to hillsides rather than to channels will also reduce the hydrologic impacts of roads (Elliot and Tysdal 1999).

Reliability and Limitations of Findings

Generally, roads will have the same types of impacts on hydrology regardless of climatic or soil differences, but the magnitude of impact may vary substantially (Elliot and others 1999a). Impacts of disturbances and benefits of mitigation measures will be greater in wetter climates. Interception of subsurface flows depends on slope position, depth to the water table, and availability of subsurface flow. The greatest challenge in applying the hydrologic findings is that landscapes are highly variable, making differences in hydrology due to the presence of roads difficult to isolate.

Research Need

The main research need is watershed scale studies to compare relatively undeveloped watersheds to similar watersheds with greater disturbances due to roads. Such sites are difficult to find, so hydrologic predictive models need to be developed and verified.

Key Point

Roads in a watershed may increase the amount of runoff and the peak runoff rate.

Sedimentation

Issues and Risks

On most forested watersheds, sediment is the most troublesome pollutant and roads are a major source of that sediment (Appelboom and others 1998; Megahan and Kidd 1972a, 1972b; Patric 1976; Reid and Dunne 1984; Yoho 1980). Sediment can adversely impact water quality by increasing turbidity, prematurely plugging filters and other components of treatment systems. Suspended sediment can also carry undesirable chemical pollutants, such as phosphates, pesticides, and other hydrocarbons into surface water and ground water (Gilson and others 1994, Patric 1976, Thomson and others 1997). See chapter 3 for additional impacts of sediment.

Sediment may be from surface erosion, which is generally more likely to carry pollutants. On steep watersheds, more sediment may be from mass wasting, which tends to bring greater volumes of soil to the stream.

Findings from Studies

Numerous researchers and managers throughout the United States have identified roads as a major source of sediment in otherwise relatively undisturbed watersheds, such as forests and rangelands (table 9.1). Table 9.1 presents some typical erosion rates for different regions in the United States for different types of disturbance. Note that some investigators have reported erosion rates for roads, ranging from 5 to 550 tons per acre [11.2 to 1232 metric tonnes (Mg) per hectare] per year, whereas others have reported erosion rates of watersheds containing roads in the range of 0.02 to 2 tons per acre (0.045 to 4.5 Mg per hectare) per year. The wide range results from differences in measuring erosion (at the road or at the watershed outlet) and in the factors causing erosion, including the presence, density, and design of the road network on the watershed.

In a mixed rural and urban watershed in northern Idaho, roads covered only 1 percent of a watershed, but they contributed 8 percent of the sediment to streams (Idaho Division of Environmental Quality 1997). Megahan (1974) estimated that, in central Idaho, the sediment yield from watersheds without roads was about 0.07 tons per square mile (0.025 Mg per square kilometer) per day, whereas the presence of roads increased this yield by a factor of 5. McNulty and others (1995) attributed the majority of sediment from a forested watershed in the Southeast to unpaved roads.

Immediately after roads are constructed, erosion rates from bare slopes and road surfaces are high (fig. 9.2). Erosion rates can drop rapidly as exposed slopes revegetate and stabilize. Erosion reductions of 90 percent or more are common as a road ages (Burroughs and King 1989, Ketcheson and Megahan 1996). Road surfaces, however, will likely continue to be a source of sediment as long as traffic or maintenance prevents the establishment of vegetation (Elliot and others 1996, Swift 1984b). Applications of high-quality gravel to unpaved roads can decrease erosion rates by up to 80 percent (fig. 9.2) (Burroughs and King 1989, Swift 1984a), but reductions may be less for poorer quality aggregates (Foltz and Truebe 1995).

In a study attempting to isolate the specific sources of sediment, Burroughs and King (1989) identified the cutslope, the roadway, and the fillslope (fig. 9.1). For each of these components they suggested mitigation measures, including application of mulch, geotextiles, seed, and sod. Many other studies have demonstrated the effectiveness of these treatments (table 9.2), and they are recommended in many States. Luce and Black (1999), however, were not able to measure any differences in sediment from roads for bare and vegetated cutslopes of different heights in the Oregon Coast Range. They concluded that the roadway and the road ditch were the only significant sources of sediment.

Wemple and others (1996) and Elliot and Tysdal (1999) found that the roads can influence a wider zone of erosion than previously thought. Slopes and channels downhill from the road can be sites of deposition, or the major source of sediment from a given segment of road. The excess runoff from roads can overload ephemeral channels, resulting in significant downcutting of the channel.

Poor road drainage can also lead to saturation of road beds and mass failure. In steep terrain, abandoned roads that do not shed surface water can become saturated, increasing the likelihood of failure. In areas of high rainfall, such as the Coast Range in Washington and Oregon, more sediment comes from roads due to landslides associated with roads than from road surface erosion. Beschta (1978) reported that watershed sediment yields increased from around 300 tons per square mile [105 Mg per square kilometer (km)] per year before roads and harvesting, to about 400 tons per square mile (140 Mg per square km) per year after installing roads and harvesting timber. Much of the increase in watershed sediment yield in this high-rainfall area was from mass failure. In a recent study in the Clearwater National Forest in Idaho, 58 percent of the landslides that occurred were associated with roads (McClelland and others 1998). Recent studies in Oregon, however, suggest that road impacts may have been overestimated (Robinson and others 1999), and that sediment from landslides in undisturbed areas is similar to that in areas with roads. While surface erosion is a chronic source of sediment associated with numerous precipitation or snowmelt events every year, landslides tend to contribute large amounts of sediment during very wet years and no sediment during normal and dry years. Landslide scars can also be sources of sediment until they are revegetated. McClelland and others (1998) calculated that the amount of sediment from the worst landslides in 20 years was about 10 times a background erosion rate, while the ongoing contribution from roads in the basin was about 2.5 times the background rate.

In addition to roads, other rights-of-way such as pipelines, are potential sources of sediment (Gray and Garcia-Lopez 1994, Sonett 1999). Any construction that exposes bare mineral soil, particularly on sites that are adjacent to ditches or streams, is likely to increase sedimentation. Once

Table 9.1—Typical erosion rates observed for different types of land use in the United States

Location	Surface cover	Erosion rate	Reference
		Tons/ac/yr	
Eastern watersheds	Forests	0.003– 0.32	Patric 1976
Fernow NF, West Virginia	Observed bare and graveled roads	6.0 – 52.5	Kochenderfer and Helvey 1987
Appalachian Trail	Trail	4 – 60	Burde and Renfro 1986
Southeast	Roads	5 –144	Swift 1984a, 1984b
Southern watersheds	Forests	Trace – .32	Yoho 1980
	Meadow	.06 – .1	
	Prescribed burn	.01 – .23	
	Careless clearcut	1.35	
	Roads	16 –150	
Central Arkansas	Roads	6.8 – 33.7	Beasley and others 1984
		4 – 38.5	Miller and others 1985
Southeastern Oklahoma	Roads	8 – 77	Vowell 1985
Western watersheds	Rangeland	.1 – 1.8	U.S. Department of Agriculture 1989
Northern Rockies	Forests	.04	Megahan 1974, McClelland and others 1998
	Forested watershed		
	Undisturbed	0	Megahan and Kidd 1972b
	With roads	.02	
	Roads	7.5 – 22	Ketcheson and Megahan 1996, Megahan and Seyedbagheri 1986, Megahan and Kidd 1972a
Washington Olympics	Roads	46 –550	Reid and Dunne 1984
Oregon Cascades	Forested watershed	.11	Fredrikson 1970 (most of road and harvest erosion attributed to landslides)
	Roads added	.56	
	Harvested	18.4	
	Roads	.22 – 24	Foltz 1996
Oregon Coast Range	Roads	1 – 18	Luce and Black 1999
Oregon coast	Forests	.4	Beschta 1978
Northern California Coast Range watershed	Undisturbed forest	.008	Rice and others 1979
	After roads	.63	
	After roads and logging	1.9	

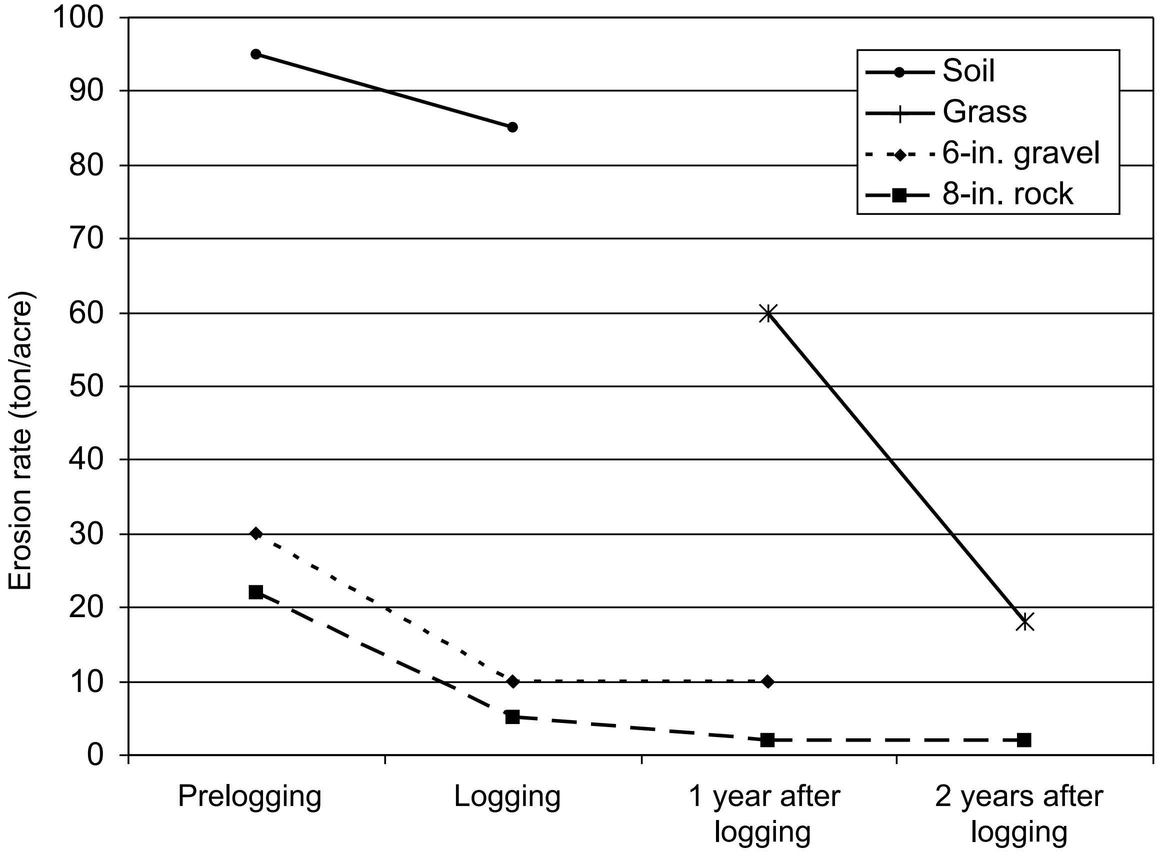


Figure 9.2—Mean soil loss rates for four road surfaces before, during, and for 2 years after logging (based on Swift 1984a).

installed, rights-of-way may continue to be sources of sediment if revegetation or other erosion control practices are not initiated (Gray and Garcia-Lopez 1994). Frequently, off-road vehicle enthusiasts may use rights-of-way for recreation. Also, mechanical and chemical control of vegetation may reduce vegetative cover. Depending on the site conditions, erosion rates from the compacted trails or exposed rights-of-way may be similar to those of roads.

Trails for bicycling, walking, or horseback riding erode at rates similar to roads (Leung and Marion 1996). The total sediment delivered from these trails is generally lower, however, because the total surface area of a narrow trail is less than that of most roads.

Much of the sediment eroded from a right-of-way is rapidly deposited below the right-of-way and never reaches a stream. Rummer and others (1997) found no significant sedimentation effects beyond the clearing limit of the road in a bottomland hardwood study on a floodplain. Numerous scientists have developed equations from field observations to predict how far sediment will travel (Ketcheson and Megahan 1996, McNulty and others 1995, Packer and Christensen 1977, Swift 1986).

Various mitigation measures to reduce road erosion are commonly prescribed by Federal and State agencies. The most common methods include surfacing the road with gravel, decreasing the spacing of cross drainage, locating roads farther from streams, or limiting road gradients (Burroughs and King 1989, Swift 1984a, Yoho 1980).

Table 9.2—Effectiveness of erosion mitigation techniques

Condition	Reduction	Reference
	Percent	
Erosion mat	74– 99	Grace and others 1998
Seeding	82– 95	Grace and others 1998
Grass on fillslope	46– 81	Appelboom and others 1998
Straw and asphalt tack or erosion mats (depends on percent cover)	60– 100	Burroughs and King 1989
Straw	60– 80	Burroughs and King 1989

Treatment of cut and fillslopes has also been effective in reducing sediment delivery from new roads (table 9.2) (Burroughs and King 1989, Grace and others 1998). Sediment production can be reduced by applying higher quality gravel (Foltz and Truebe 1995) or by reducing the pressure in vehicle tires on the road network (Foltz 1994). The installation of vegetated filter strips or slash filter windrows below fills, or sediment basins below culverts, are also management options that have reduced sedimentation. In climates with distinct wet seasons, seasonal closure of roads may be a desirable option to prevent rutting and severe erosion.

In an effort to reduce the impacts of roads and railroads in watersheds, many government management agencies are removing unwanted corridors. In many national forests, watershed restoration is synonymous with removal of excess roads (Elliot and others 1996). Moll (1996) presented an overview of road closure and obliteration methods in the Forest Service. He recommended that watershed managers consider access, drainage, erosion risk, slope stability, and revegetation when planning any road closure or obliteration. Table 9.3 summarizes management options for decommissioned roads. Elliot and others (1996) warn that the disturbances associated with road closure may cause more erosion than simply abandoning a road that has been revegetated and is hydrologically stable.

Surface erosion rates can drop significantly when roads are closed. Figure 9.2 shows the relative impacts of different road surfaces during the first 2 years after abandonment, compared to erosion rates during construction and logging (Swift 1984b). In the Oregon Cascades, Foltz (1996) observed that during the first year of closure erosion rates dropped from 4 to 0.5 tons per acre (9 to 1 Mg per hectare) when marginal quality aggregate was applied, 20 to 2.5 tons per acre (45 to 5.6 Mg per hectare) when good-quality aggregate was applied. Erosion will often drop to background levels as the density of vegetation on an abandoned road surface increases (Foltz 1996, Swift 1984b). Such a decline is unlikely, however, if the abandoned road has unvegetated surfaces and continues to concentrate runoff water.

Several general principles can be applied to analyzing and mitigating potential sediment sources from abandoned corridors. The surface should be covered with vegetation. In order to establish vegetation, it may be necessary to rip or till the surface. In extreme cases, it may be necessary to add topsoil. To encourage infiltration and revegetation, it may be necessary to discourage off-road vehicle traffic by installing permanent barriers to prevent wheeled access to the corridor.

On abandoned roads, culverts can fail or become blocked, causing ponding of water, embankment failure, and major offsite sedimentation (Elliot and others 1994). Many older roads or railroads were built with underdesigned culverts. Some culverts were made from wood that is decaying or metal that is corroding. In either case, most of these culverts will eventually fail unless they are removed or replaced. Culverts that are not regularly inspected can also become blocked with woody debris or sediment. One of the most common practices to minimize the risk of fill failure on abandoned rights-of-way is to remove the culverts.

A number of prediction models have been developed to estimate the amount of sediment that leaves forest roads. Site-specific models were developed in the northern Rocky Mountains by Forest Service hydrologists; the most recent is the Watershed and Sediment Yield Model (WATSED) model (U.S. Department of Agriculture, Forest Service 1990). McNulty and others (1995) presented a Geographical Information System-based method for predicting sediment delivery from a road network, but they observed that additional work with physically based models is necessary to improve the prediction of sediment delivery from roads.

The physically based Water Erosion Prediction Project (WEPP) model is under development for a wide range of conditions including agriculture, range, and forest conditions (Laflen and others 1997). Because it is physically based, the model can be applied whenever the factors that cause erosion can be adequately described. Elliot and Hall (1997) have developed a set of input templates for forest roads and other disturbances. Elliot and others (1999b) developed simplified tools based on the WEPP model to aid managers in estimating the impacts of climate, soil, and topography on the delivery of sediment from roads. These models are available on the Web at http://forest.moscowfsl.wsu.edu/fswepp/.

Reliability and Limitations of Findings

Researchers worldwide have measured increased sedimentation from roads and similar disturbances. The magnitude of erosion varies considerably with climate, but the relative impacts of soil, topography, and management are generally the same (Elliot and others 1999b). Observed erosion rates are highly variable (table 9.1) due to the high natural variability in the factors that cause erosion. Even a well-designed erosion experiment frequently results in variations from the mean of up to 50 percent. This high variability should be considered when interpreting any research or monitoring results, or any erosion prediction value. Managers should exercise caution when applying any model to an area where it has not received some validation. Predictive technology for one climate, soil, and topography does not

Table 9.3—Management options for decommissioned roads

Option	Comment
Close road with barriers, vegetation, ditch, or removal of first segment	Recreational users may still obtain access.
Rip road surface	Runoff is reduced (Luce 1997). Instability may be increased (Elliot and others 1996).
Revegetate road surface	See table 9.2
Remove culverts and restore channels	Mitigation may be necessary on bare, excavated embankments or in channels; remaining road segments may not be accessible for future maintenance (Moll 1996).
Reshape road surface to be outsloping or partially recontoured with regular waterbars	Moll 1996
Install rock buttresses to stabilize cut and fillslopes	Moll 1996
Remove, recontour, or obliterate road prism	Expensive [$0.60 to 1.50 per lineal ft (Moll 1996)] Increase revegetation rate by excavating until the old topsoil is reached[a]
Mitigate obliterated road prism with slash, mulch, geotextile, or seeding	Moll 1996

[a] Spreiter, T.A. 1999. Road closure experiences of the National Park Service. Presented at the Oregon and Washington engineering workshop; 1999 March 9; Gleneden Beach, OR. Oral presentation.

translate well to other conditions unless the model is able to incorporate those site-specific characteristics.

The technology to remove abandoned roads is well established (Moll 1996). Numerous agencies including the Forest Service (Moll 1996; U.S. Department of Agriculture, Forest Service 1998) and the National Park Service[2] have specialists to provide technical assistance in road closure, stabilization, revegetation, and removal. Many abandoned roads and railroads require site-specific prescriptions for reclamation. The same level of design that went into creating some of these roads may be required to remove them (Elliot and others 1996). Although this design expertise is available, the cost may be prohibitive.

[2] Spreiter, T.A. 1999. Road closure experiences of the National Park Service. Presented at the Oregon and Washington engineering workshop; 1999 March 9; Gleneden Beach, OR. Oral presentation.

Research Needs

1. Upland erosion and sedimentation are well understood. The long-term impacts of trapping sediment on hillsides between sites of disturbance and streams and movement of sediment within and through stream networks are not well understood. Future research on overland transport and storage of sediment and transport and storage in stream networks will enhance sedimentation prediction.
2. Reports related to problems associated with abandoned roads and railroads focus specifically on culverts or mass failures. Surface and channel erosion may be a chronic source of sediment for many years. Thus, published information is frequently limited to episodic problems rather than solutions to chronic problems. There is a need for research to determine the probability of a failure occurring as well as the probability that no failure will occur.

3. Research is also needed to determine risks of failure or erosion for specific road networks.
4. There is a need to develop field techniques to assist road and watershed managers to make better decisions on which segments of a road network are at the greatest risk of a failure that may impact off-site water quality as well as other resources.
5. Another need is to develop tools to estimate the amount of sediment that may come from road closure activities, both from reshaping or removing the road prism, and from removal of stream-crossing structures.

Key Points

1. Roads and similar corridors can be a major source of sediment in a forested watershed.
2. Effective measures to reduce sedimentation include surface gravel, careful design of roads and water crossings, and removal of unwanted roads.
3. Abandoned roads may be sources of sediment if they collect or divert surface runoff.

Hydrocarbons, Cations, and Related Pollutants

Issues and Risks

Runoff from roads and similarly surfaced sites can contain a host of hydrocarbons and other chemical pollutants, adsorbed to sediments, as particles, or dissolved by the runoff. These chemicals can find their way into surface and subsurface water. Pesticides used to control unwanted weeds can also be a source of pollution, and the reader should refer to chapter 12 for further discussion.

Findings from Studies

Researchers have identified a range of chemicals in road runoff (tables 9.4, 9.5). Some of the pollutants are from the road material itself, some occur in the soil and rock on the site and are released during construction or subsequent erosion, and many are from vehicles. Traffic and road surfacing may contribute undesirable cations, hydrocarbons, and metals to surface and subsurface water (Maltby and others 1995, Mungur and others 1994). Most studies on the impact of roads and similar disturbances have focused on heavily traveled roads such as major freeways (Mungur and others 1994). If water source areas contain major roads, runoff treatment may be necessary to ensure that undesirable hydrocarbons do not enter the water supply.

Cations released from a road may have a buffering effect on the runoff acidity, which may be beneficial in acid rain areas. Morrison and others (1995) measured pH values from 6.0 to 7.0 in road runoff from small storms, compared to the average rainfall pH of 4.1.

Ions from deicing or dust control chemicals are common pollutants from road surfaces (Church and Friesz 1993, Pugh and others 1996). Road salt contamination of shallow ground water has become a serious problem, particularly in the Northeast and Midwest (Church and Friesz 1993). Church and Friesz (1993) state that during a 7-year period in Massachusetts, there were complaints from 100 of the 341 municipalities about road salt contamination. Nationally, about $10 million are spent each year for prevention or remediation of problems associated with road salt contamination. Surface water is less vulnerable to such contamination than ground water, because there tends to be much greater dilution and mixing in turbulent channels carrying runoff from roads (Jongedyk and Bank 1999). Calcium magnesium acetate and potassium acetate are deicing chemicals with less serious environmental consequences than sodium chloride because they contain weak biodegradable acids. Sodium chloride, calcium chloride, and magnesium chloride, however, leave residues of chloride ions that may contaminate ground water (Jongedyk and Bank 1999).

Some of these ions (calcium, magnesium, and potassium) can enhance vegetation growth along highways (Pugh and others 1996). In some cases, elevated levels of deicing cations such as sodium in the road runoff, may be adsorbed by the soils near the road, and pose no further concern to the aquatic ecosystem (Shanley 1994). Pugh and others (1996) observed that ion concentrations from an adjacent interstate highway decreased exponentially with distance from the road in a peat bog. Although many thousands of tons of salt are spread annually on highways, because of dilution, salts in runoff are not likely to be a major source of pollution for drinking water except where they use shallow ground water even though impacts on the aquatic ecosystem may be great.

Road dust can transport unwanted chemicals to surface water. Christensen and others (1997) observed recent accumulations of polycyclic aromatic hydrocarbons (PAH's) in a Wisconsin stream, and identified dust from nearby roads as the source of the pollutant.

Oil-based dust suppressants may be environmentally more risky than salt-based products. A literature search for the Forest Service (Heffner 1997) found reports that calcium and magnesium chloride showed some toxicity towards

Table 9.4—Pollutants that have been observed in runoff from road surfaces

Pollutant	Comment	Reference
Cd, Cu, Pb, Zn	Treated in wetlands	Mungur and others 1994
Highway deicing salt	Na adsorbed in soil	Shanley 1994
Cu, Zn, hydrocarbons, PAH's	Accumulated in aquatic biota	Maltby and others 1995
PAH's	Altered aquatic communities	Boxall and Maltby 1997
Ca, Mg, Na, K, Cl	Captured in peat bogs	Pugh and others 1996
Total petroleum hydrocarbons, Pb, Zn	Reduced by vegetation	Ellis and others 1994
Heavy metals, petroleum hydrocarbons, pesticides, sediment, nutrients	Treatment ponds can remove up to 95 percent of pollutants.	Karouna-Renier and Sparling 1997

Ca = calcium; Cd = cadmium; Cl = chloride; Cu = copper; K = potassium; Mg = magnesium; Na = sodium; PAH's = polycyclic aromatic hydrocarbons; Pb = lead; Zn = zinc.

plants, whereas ligninsulfonate increased water biological oxygen demand. The study concluded:

> based on the literature review and typical application rates for dust abatement, the effects of these compounds on plants and animals would be negligible. For the purposes that the Forest Service uses these compounds, the selection of one over another would be more dependent on cost, availability, and local conditions than effects to the environment.

Some dust inhibitors may also decrease road erosion, decreasing the likelihood of off-site transport of sediment and related pollutants (Ice 1982).

Chemicals used to preserve utility poles and railway crossties are potential sources of pollution. Wan (1994) found that concentrations of PAH's in the soil were higher in the immediate vicinity of utility poles than on surrounding farm or rangeland. Soil concentrations of PAH's dropped rapidly from 550 micrograms (µg) per liter to 23.2 µg per liter within 13 feet (4 meters) of a treated pole. Background levels were between zero and 0.8 µg per liter. Such findings emphasize the importance of maintaining vegetated buffers to reduce transport by erosion of contaminated soil between rights-of-way and any sensitive water resources.

Measuring concentrations of many pollutants is tedious and expensive. To reduce the cost, surrogate relationships have been developed between more easily measured pollutants, such as suspended solids (mainly sediment) or dissolved

Table 9.5—Mean concentrations of a number of pollutants in highway runoff in Minnesota

Pollutant	Range	Mean
	Milligrams per liter	
Total nitrogen	0.6 – 8.14	1.67
Chloride	1 – 46 000	1 802
Sulphate	5 – 650	45
Sodium	2 – 67 000	3 033
Total phosphorous	.06 – 7.8	.6
BOD	1 – 60	12.6
COD	2 – 3 380	207
Total suspended solids	8 – 950	118
Total dissolved solids	22 – 81 700	10 440
	Micrograms per liter	
Chromium	1.5 – 110	13
Copper	3 – 780	47
Iron	180 – 45 000	4 162
Lead	11 – 2 100	207
Zinc	10 – 1 200	174
Nickel	1 – 57	10
Cadmium	.2 – 12	1.7
Mercury	.08 – 5.6	.49
Aluminum	30 – 14 000	2 694
Arsenic	.1 – 340	19

BOD = biological oxygen demand; COD = chemical oxygen demand.
Source: Thomson and others 1997.

solids and other pollutants that are difficult to measure (Thomson and others 1997). Such surrogates may be useful if relationships were developed for nearby conditions, but they become less reliable when extrapolated to other regions.

Gilson and others (1994) completed research on the effectiveness of filter systems for highway runoff to improve surface water quality in the karst terrain in Texas. They found that some alternatives to sand filters have higher adsorptive capacities initially, but filtration efficiencies tended to approach that of sand after several runoff events. A Virginia study found that highways in karst areas should be located to avoid polluting surface water that drains into caves (Hubbard and Balfour 1993). This study found raw sewage and petroleum fumes in the cave system. Keith (1996) described extra precautions on road location and drainage designs that were taken in Indiana to minimize the ground water impact of a new road design in a karst area.

Pollutants in runoff can be trapped in natural or artificial wetland areas (Ellis and others 1994, Karouna-Renier and Sparling 1997, Mungur and others 1994). Karouna-Renier and Sparling (1997) found that such treatment systems could remove up to 95 percent of metals, nutrients, and sediment. Monitoring of the performance of such areas is necessary to ensure that they are functioning as desired (Startin and Lansdown 1994).

Another treatment method is a partial exfiltration trench. This type of device filters out the suspended solids that carry many of the undesirable metals and hydrocarbons from road surfaces (Sansalone and Buchberger 1995). The trench improved the quality of both rainfall and snowmelt runoff from roads. Because of the wide range of runoff rates, however, multiple treatment methods may be necessary to decrease the pollutant load from large as well as small storms (Romero-Lozano 1995). Detention basins are needed to catch the first flush of highly polluted runoff. A filtration system is needed to treat the runoff from later in the storm, which is likely to be at a higher flow rate, but requires less treatment.

Sediment basins and similar structures built to contain polluted road runoff can become sources of pollution through seepage into the ground, or through other forms of hydraulic or structural failure. In either case, sediments with large amounts of adsorbed chemicals can enter a stream. The pollutants can become concentrated in these basins, increasing the risk of offsite pollution (Grasso and others 1997, Morrison and others 1995). Grasso and others (1997) observed a lead content of 1392 milligrams per kilogram on one site and recommended soil washing be carried out to prevent offsite pollution. One of the best defenses against such risks is cleaning and maintaining such structures to minimize the risk of failure.

Past designs of runoff structures tended to collect water and route it directly to a stream. New designs that disperse water to ensure greater infiltration and onsite attenuation of pollutants can improve runoff quality (Elliot and Tysdal 1999, Li and others 1998). Not all sites lend themselves to this approach, particularly where rights-of-way are limited. Another recent innovation to reduce offsite pollution from roads and similar areas is to surface them with permeable pavement (Church and Friesz 1993). Permeable pavement combined with high-infiltration shoulders significantly decreased salt content in nearby ground water (Church and Friesz 1993, Jongedyk and Bank 1999). European researchers found that permeable pavement significantly reduced outflow levels of lead and suspended solids.

Reliability and Limitations of Findings

Much of the research associated with chemical pollution from roads has taken place near large urban centers. The findings are generally reliable for their locality, but care needs to be taken in extrapolating to other conditions, particularly nonurban areas. The water-quality risks associated with hydrocarbon pollution are closely linked to the density of traffic. Watersheds with minimal traffic are unlikely to experience any of the problems discussed. These results should only be applied to more remote watersheds with caution and some form of monitoring.

Research Need

Pollution from main roads that cross sensitive forest and grassland watersheds should be measured. Quantitative data are needed on the benefits of dust abatement chemicals for reducing erosion and pollution of streams near roads.

Key Point

Many pollutants from vehicles, deicing and dust abatement chemicals, and road surfacing material have been measured in runoff from roads. Most of these measurements have been from roads with heavy traffic. Some level of monitoring may be necessary to determine pollution problems. Levels of pollution can often be related to levels of easily measured sediment concentration. Some cations in runoff may be beneficial in buffering acid rain. There are methods to collect and treat or to harmlessly disperse polluted road runoff.

Fuels and Other Contaminants from Accidental Spills

Issues and Risks

Accidents are rare on low-use roads and other rights-of-way in remote watersheds. Risks of an accident causing contamination spills are related to the traffic density, quality of road, and frequency of contaminant transport. Railroads pose similar risks, particularly on aging lines, or on busy routes linking industrial centers.

Findings from Studies

Hazardous chemical spills from vehicle accidents can pose a direct, acute threat of contamination to streams. Risk analysis models have been developed for busy paved roads in nonmountainous terrain, but these models are seldom applicable to low traffic, remote watersheds. Chemicals that may be spilled include fuel, fertilizer, pesticides, and mining chemicals (U.S. Department of Agriculture, Forest Service 1998). Airfields can often be sources of ground water contamination due to spills of fuels and other material (Levine and others 1997).

Accidents may occur anywhere along a given road or railroad, but stream crossings and bridges tend to be frequent sites of accidents due to damage by floods, or a narrowing of the roadway. Whether the pollutant is able to reach a nearby stream is an important concern. Spills at stream crossings have a high likelihood of reaching surface water because of its close proximity. Frequently, transport of the pollutant overland, or through the soil, depends on the local climate, season, and hydrology.

Reliability and Limitations of Findings

There is little information available on the risk of accidental spills in remote areas. Whatever information can be found is likely to be site-specific, and judgment must be used to apply it elsewhere. Watershed managers will need to develop their own set of potential risks, based on local conditions. Along with those risks, they will need to develop a set of potential mitigation measures, both in the water source area, and in the treatment system.

Ability to Address Issues

Most counties have established committees to address local emergencies or disasters. An accident that impacts a local water supply is a prime example of such an emergency. Water supply managers should work with local emergency or disaster committees or services to ensure that mitigation plans and equipment are in place to deal with toxic spills that may occur near a water source.

Research Need

Because of the site-specific nature of this risk, it is difficult to define a broad research activity for remote watersheds. It is likely that research will continue to study risks associated with busier roads, so monitoring of those results for application to remote watersheds may be beneficial.

Key Point

The risk of vehicle accidents and spills depends on road hazards and traffic volume. Watershed managers need to evaluate risks on a given watershed and develop prevention or mitigation measures specific to their own conditions.

Pipeline Failures

Issues and Risks

Pipelines carrying a wide range of substances, including drinking water, sewage, and petroleum products, can fail, leading to pollution of both surface water and ground water. In the past 15 years, about 200 oil pipeline failures have occurred per year, with an average net loss of about 600 barrels (95 cubic meters) for each spill (U.S. Department of Transportation 1999).

Findings from Studies

Pipelines tend to have fewer accidents and injuries than other modes of transport (Jones and Wishart 1996). To minimize pollution impacts, most modern pipeline systems are equipped with devices to quickly shut down pumping if a change in flow or pressure is sensed (Ariman 1990).

A number of disturbances increase the likelihood of pipeline accidents (fig. 9.3). Road or construction accidents and damage from boulders are common external causes of damaged pipelines (Driver and Zimmerman 1998, Stalder 1997). Areas prone to severe erosion, landslides, and earthquakes tend to have more accidents (Ariman 1990, Gray and Garcia-Lopez 1994, Hart and others 1995). For example, Hart and others (1995) predicted that the probability of rupture for a pipeline in California increased from 0.0 for earthquakes with a magnitude below 5 to 1.0 for earthquake magnitudes greater than 6.0. They also predicted other probabilities of failure based on pipe length and installation. They recommended numerous design measures including depth of burial, trench design, and pipe wall thickness, to minimize failure due to earthquakes.

Pipelines sometimes fail at river crossings due to erosion of the streambank or bottom (Doeing and others 1995, Teal and others 1995). Pipelines carrying sewage and industrial wastes are frequently located in floodplains and are at particular risk from flood damage, or from overloading due to high runoff rates. Disturbances in a watershed, such as a fire, may cause landslides that lead to pipeline failure. Failure of water supply pipelines or canals can lead to considerable erosion if controls to monitor flow conditions are not in place.

Soil shrinking and swelling and freezing and thawing can lead to pipeline fatigue and premature failure. Corrosion due to electrolysis (Stalder 1997) and stress corrosion cracking can also occur on older pipelines (Wilson 1996). Above-ground pipelines can fail due to wind fatigue (Honegger and others 1985). Any pipeline may experience seam failure (Yaorong and others 1996).

Risk assessment models to aid in pipeline design and operation have been developed (Hart and others 1995, Nessim and Stephens 1998). These models can identify segments of pipe most at risk of failure, and maintenance can be concentrated on those segments. Risk rates of 0.0022 spills per mile per year are quoted in one environmental assessment (U.S. Department of the Interior, Bureau of Land Management 1978).

Table 9.6 shows the extent of contamination from 53 oil pipeline spills. The extent depends on the pipeline characteristics and on the soils and terrain. Risks of failure from normal, predictable events can be reduced to almost zero with adequate pipeline monitoring (Stalder 1997). In addition, technologies have been developed to mitigate the impacts of spills quickly and effectively (Sittig 1978).

Pipeline failures can pollute ground water as well as surface water. Petroleum products tend to float on ground water, but the processes associated with breakdown of oil underground are not well understood. Underground methane generation by anaerobic bacteria is common after a pipeline break.

> Substantial amounts of the volatile petroleum hydrocarbons are transported from the surface of the water table through the unsaturated zone as vapor, which subsequently dissipates to the atmosphere or is biodegaded (Revesz and others 1996).

Eganhouse and others (1996) observed that an underground breakdown process from microbial degradation leads to the detection of a plume containing aliphatic, aromatic, and alicyclic hydrocarbons.

Pipelines carrying water and sewage may also be present on watersheds. Although the same principles of failure apply to these pipelines, they are generally not as well monitored and may be older and more prone to failure.

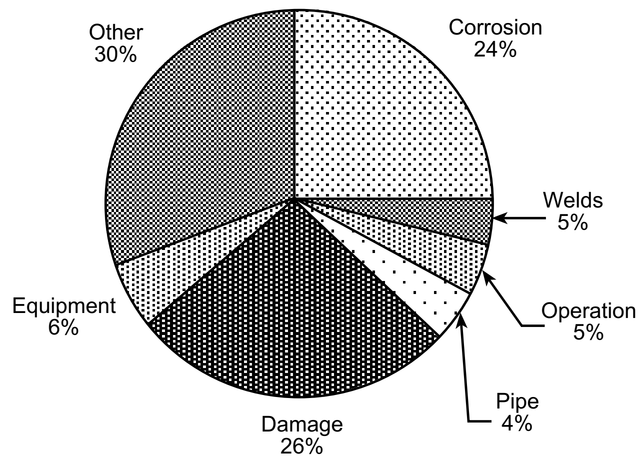


Figure 9.3—Summary of causes of pipeline accidents in 1998 in the United States (U.S. Department of Transportation 1999).

Reliability and Limitations of Findings

These findings are generally reliable because much of the pipeline industry receives close government scrutiny. Pipeline failures tend to be mechanical and predictable and findings are generally applicable to local conditions.

Current technology can address the risks associated with pipeline failure. Technologies to minimize pipeline failure are well established in the petroleum industry, as are controls to minimize pollution of surface and ground water should a failure occur. These same technologies can also be applied to other pipelines in sensitive watersheds. Managers of watersheds containing pipelines should work with

Table 9.6—Extent of soil contamination by 53 oil spills of various sizes in Alberta, Canada

Average volume	Average area	Average film thickness
Barrels	*Ft^2*	*In.*
54	8,000	0.4
880	70,000	.8
13,200	600,000	1.6

Barrel = 42 gallons of petroleum.
Source: U.S. Department of the Interior, Bureau of Land Management 1978.

pipeline managers to minimize risks to water supplies. In addition, the U.S. Department of Transportation has an Office of Pipeline Safety (OPS) to assist in addressing risks associated with pipelines. One of its responsibilities is to identify areas that are unusually sensitive to a hazardous liquid pipeline release. The OPS has an ongoing program that may assist watershed managers in risk management (see Web site).

Research Need

The oil industry has developed sophisticated systems for managing pipelines. There is a need to develop similar, but less costly, technologies for water and sewer pipelines in sensitive watersheds. The fate of oil pollution in the ground is not well understood, and further research is needed to better understand the chemical and biological processes associated with degradation of petroleum products on and in the soil.

Key Point

Causes of failures on petroleum pipelines are well understood, and controls are generally in place to minimize environmental risks of failures. Such measures are less developed for water and sewage pipelines, so the risks of the failure of such systems may be higher.

Literature Cited

Appelboom, T.W.; Chescheir, G.M.; Skaggs, R.W.; Hesterberg, D.L. 1998. Evaluating management practices for reducing sediment production from forest roads. American Society of Agricultural Engineers annual international meeting; 1998 July 12–16; Orlando, FL. Pap. 987025. St. Joseph, MI: American Society of Agricultural Engineers. 11 p.

Ariman, T. 1990. Seismic vulnerability of crude oil pipelines. Pipeline and Gas Journal. 217(July): 22–24.

Beasley, R.S.; Miller, E.L.; Gough, S.C. 1984. Forest road erosion in the Ouachita Mountains. In: Peters, P.A.; Luchok, J., eds. Mountain logging symposium proceedings; 1984 June 5–7; Morgantown, WV. Morgantown, WV: West Virginia University: 203–213.

Beschta, R.L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon coast range. Water Resources Research. 14(8): 1011–1016.

Boxall, A.B.A.; Maltby, L. 1997. The effects of motorway runoff on freshwater ecosystems: 3. Toxicant confirmation. Archives of Environmental Contamination and Toxicology. 33(1): 9–16.

Burde, J.H.; Renfro, J.R. 1986. Use impacts on the Appalachian trail. In: Lucas, R.C., comp. Proceedings—national wilderness research conference: current research; 1985 June 5–7; Fort Collins, CO. Gen. Tech. Rep. INT–212. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 138–143.

Burroughs, E.R., Jr.; King, J.G. 1989. Reduction of soil erosion on forest roads. Gen. Tech. Rep. INT–264. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 21 p.

Burroughs, E.R., Jr.; Marsden, M.A. 1972. Volume of snowmelt intercepted by logging roads. Journal of the Irrigation and Drainage Division. March: 1–12.

Christensen, E.R.; Li., A.; Razak, I.A.A. [and others]. 1997. Sources of polycyclic aromatic hydrocarbons in sediments of the Kinnickinnic River, Wisconsin. Journal of Great Lakes Research. 23(1): 61–73.

Church, P.E.; Friesz, P.J. 1993. Effectiveness of highway drainage systems in preventing road-salt contamination of groundwater: preliminary findings. Transp. Res. Rec. 1420. Washington, DC: National Academy Press: 56–60.

Doeing, B.J.; Bradley, J.B.; Carreon, S. 1995. Avoiding pipeline failures at waterway crossings. Part 1: scour assessment methods. In: Proceedings of the 2d international conference on advances in underground pipeline engineering; 1995 June 25–28; Bellvue, WA. New York: American Society of Civil Engineers: 65–76.

Driver, R.G.; Zimmerman, T.J.E. 1998. Limit States' approach to the design of pipelines for mechanical damage. In: Proceedings of the 17th international conference on offshore mechanics and arctic engineering—OMAE; 1998 July 5–9; Lisbon, Portugal. Fairfield, NJ: American Society of Mechanical Engineers: 1–14.

Eganhouse, R.P.; Dorsey, T.F.; Phinney, C.S.; Westcott, A.M. 1996. Processes affecting the fate of monoaromatic hydrocarbons in an aquifer contaminated by crude oil. Environmental Science and Technology. 30(11): 3304–3312.

Elliot, W.J.; Foltz, R.B.; Luce, C.H. 1999a. Modeling low-volume road erosion. In: Proceedings of the 7th international conference on low-volume roads; 1999 May 23–27; New Orleans. Transp. Res. Rec. 1652. Washington, DC: National Academy Press. (2): 244–250.

Elliot, W.J.; Foltz, R.B.; Luce, C.H.; Koler, T.E. 1996. Computer-aided risk analysis in road decommissioning. In: McDonnell, J.J.; Stribling, J.B.; Neville, L.R.; Leopold, D.J., eds. Proceedings of the AWRA annual symposium on watershed restoration management: physical, chemical, and biological considerations; 1996 July 14–17; Syracuse, NY. Herndon, VA: American Water Resources Association: 341–350.

Elliot, W.J.; Hall, D.E. 1997. Water erosion prediction project (WEPP) forest applications. Gen. Tech. Rep. INT–365. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 11 p.

Elliot, W.J.; Hall, D.E.; Graves, S.R. 1999b. Predicting sedimentation from forest roads. Journal of Forestry. 97(8): 23–29.

Elliot, W.J.; Koler, T.E.; Cloyd, J.C.; Philbin, M. 1994. Impacts of landslides on an ecosystem. Pap. 947517. Presented at the 1994 American Society of Agricultural Engineers international winter meeting; 1994 December 13–16; Atlanta. St. Joseph, MI: American Society of Agricultural Engineers. 10 p.

Elliot, W.J.; Tysdal, L.M. 1999. Understanding and reducing erosion from insloping roads. Journal of Forestry. 97(8): 30–34.

Ellis, J.B.; Revitt, D.M.; Shutes, R.B.E.; Langley, J.M. 1994. The performance of vegetated biofilters for highway runoff control. In: Hamilton, R.S.; Revitt, D.M.; Harrison, R.M.; Monzon-de-Caceres, A., eds. Proceedings of the 4th international symposium on highway pollution; 1992 May 18–22; Madrid, Spain. London: Middlesex University: 543–550.

Foltz, R.B. 1994. Sediment reduction from the use of lowered tire pressures. Central tire inflation systems—managing the vehicle to surface (SP–1061). Pap. 942244. Warrendale, PA: Society of Automotive Engineers: 47–52.

Foltz, R.B. 1996. Traffic and no-traffic on an aggregate surfaced road: sediment production differences. In: Proceedings of the seminar of environmentally sound forest roads and wood transport; 1996 June 17–22; Sinaia, Romania. Rome: Food and Agriculture Organization of the United Nations: 195–204.

Foltz, R.B.; Truebe, M.A. 1995. Effect of aggregate quality on sediment production from a forest road. In: Proceedings of the 6th international conference on low-volume roads; 1995 June 25–29; Minneapolis. Washington, DC: National Academy Press: 49–57.

Fredrickson, R.L. 1970. Erosion and sedimentation following road construction and timber harvest on unstable soils in three small western Oregon watersheds. Res. Pap. PNW–104. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 15 p.

Gilson, M.P.; Maliva, J.F.; Charbeneau, R.J.; Ward, G.H. 1994. Highway runoff studied. Water Environment & Technology. 6(8): 37–38.

Grace, J.M., III; Rummer, R.; Stokes, B.; Wilhoit, J. 1998. Evaluation of erosion control techniques on forest roads. Transactions of the American Society of Agricultural Engineers. 4(2): 383–391.

Grasso, D.; Butkus, M.A.; O'Sullivan, D.; Nikolaidis, N.P. 1997. Soil-washing design methodology for a lead-contaminated sandy-soil. Water Research. 31(12): 3045–3056.

Gray, D.H.; Garcia-Lopez, M. 1994. Protecting a pipeline. Civil Engineering. 64(12): 44–47.

Hart, J.D.; Row, D.G.; Drugovich, G. 1995. Structural analysis and risk assessment of the All American pipeline at the San Andreas fault crossing. In: American Society of Mechanical Engineers, petroleum division PD pipeline engineering 1995: Proceedings of the energy-sources technology conference and exhibition; 1995 January 29–February 1; Houston. New York: American Society of Mechanical Engineers: 1–15.

Heffner, K. 1997. Water quality effects of three dust-abatement compounds. U.S. Department of Agriculture, Forest Service. Engineering Field Notes. 29: 35–43.

Honegger, D.G.; Nyman, D.J.; Nyman, K.J. 1985. Wind induced fatigue damage estimated for surface oil pipelines in the Arctic. Oil and Gas Journal. 83(July): 49–53.

Hubbard, D.A., Jr.; Balfour, W.M. 1993. An investigation of engineering and environmental concerns relating to proposed highway construction in a karst terrain. Environmental Geology. 22(4): 326–329.

Ice, G. 1982. An assessment of the use of lignosulfonates and possible water quality impacts to forest streams. Corvallis, OR: National Council of the Paper Industry for Air and Stream Improvement. 3 p.

Idaho Division of Environmental Quality. 1997. Paradise Creek TMDL water body assessment and total maximum daily load. Lewiston, ID: Idaho Division of Environmental Quality, Lewiston Regional Office. 72 p.

Jones, E.G.; Wishart, D.M. 1996. Pipelining in a changing competitive environment. In: Proceedings of the 1st international pipeline conference; 1996 June 9–13; Calgary, Canada. New York: American Society of Mechanical Engineers: 55–62.

Jones, J.A.; Grant, G.E. 1996. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. Water Resources Research. 32(4): 959–974.

Jones, J.A., Swanson, F.J.; Wemple, B.C.; Snyder, K.U. [In press]. Road and stream network interactive effects on hydrology and geomorphology in western Oregon. Conservation Biology.

Jongedyk, H.; Bank, F. 1999. Is highway runoff a serious problem? Federal Highway Administration Environmental Technology Brief. McLean, VA: U.S. Department of Transportation. 9 p. [Available on the Web at: http://www.tfhrc.gov/hnr20/runoff/runoff.htm].

Karouna-Renier, N.K.; Sparling, D.W. 1997. Toxicity of stormwater treatment pond sediments to *Hyalella azteca* (Amphipoda). Bulletin of Environmental Contamination and Toxicology. 58(4): 550–557.

Keith, J.H. 1996. Runoff treatment systems reduce highway-related impacts in the Indiana karst. Land-water. 40(3): 12–16.

Ketcheson, G.L.; Megahan, W.F. 1996. Sediment production and downslope sediment transport from forest roads in granitic watersheds. Res. Pap. INT–486. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 11 p.

Kochenderfer, J.N.; Helvey, J.D. 1987. Using gravel to reduce soil losses from minimum-standard forest roads. Journal of Soil and Water Conservation. 42(1): 46–50.

Laflen, J.M., Elliot, W.J.; Flanagan, D.C. [and others]. 1997. WEPP—Predicting water erosion using a process-based model. Journal of Soil and Water Conservation. 52(2): 96–102.

Leung, Y.; Marion, J.L. 1996. Trail degradation as influenced by environmental factors: a state-of-the-knowledge review. Journal of Soil and Water Conservation. 51(2): 130–136.

Levine, A.D.; Libelo, E.L.; Bugna, G. [and others]. 1997. Biogeochemical assessment of natural attenuation of JP–4-contaminated groundwater in the presence of fluorinated surfactants. Science of the Total Environment. 208(3): 179–195.

Li, J.; Orland, R.; Hogenbirk, T. 1998. Environmental road and lot drainage designs: alternatives to the curb-gutter-sewer system. Canadian Journal of Civil Engineering. 25(1): 26–39.

Luce, C.H. 1997. Effectiveness of road ripping in restoring infiltration capacity of forest roads. Restoration Ecology. 5(3): 265–270.

Luce, C.H.; Black, T.A. 1999. Sediment production from forest roads in western Oregon. Water Resources Research. 35(8): 2561–2570.

Maltby, L.; Boxall, A.B.A.; Forrow, D.M. [and others]. 1995. The effects of motorway runoff on freshwater ecosystems. 2. Identifying major toxicants. Environmental Toxicology and Chemistry. 14(6): 1093–1101.

McClelland, D.E.; Foltz, R.B.; Falter, C.M. [and others]. 1998. The relative effects of landslides resulting from episodic storms on a low-volume road system in northern Idaho. U.S. Department of Agriculture, Forest Service. Engineering Field Notes. 30: 7–22.

McNulty, S.; Swift, L., Jr.; Hays, J.; Clingenpeel, A. 1995. Predicting watershed erosion production and over-land sediment transport using a GIS. In: Carrying the torch for erosion control: an olympic task. Proceedings of the 26th international erosion control association conference; 1995 February 28–March 3; Atlanta. Steamboat Springs, CO: International Erosion Control Association: 397–400.

Megahan, W.F. 1974. Erosion over time on severely disturbed granitic soils: a model. Res. Pap. INT–156. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 26 p.

Megahan, W.F. 1983. Hydrologic effects of clearcutting and wildfire in steep granitic slopes in Idaho. Water Resources Research. 19(3): 811–819.

Megahan, W.F.; Kidd, W.J. 1972a. Effect of logging roads on sediment production rates in the Idaho batholith. Res. Pap. INT–123. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 14 p.

Megahan, W.F.; Kidd, W.J. 1972b. Effects of logging and logging roads on erosion and sediment deposition from steep terrain. Journal of Forestry. 70(3): 136–141.

Megahan, W.F.; Seyedbagheri, K.A. 1986. Construction phase sediment budget for forest roads on granitic slopes in Idaho. In: Drainage basin sediment delivery; Proceedings of a conference; 1986 August 4–8; Albuquerque, NM. IAHS Publ. 159; Wallingford, UK: International Association for Hydrological Sciences: 31–39.

Miller, E.L.; Beasley, R.S.; Covert, J.C. 1985. Forest road sediments: production and delivery to streams. In: Blackmon, B.G., ed. Proceedings of forestry and water quality: a mid-South symposium; 1985 May 8–9; Little Rock, AR. Fayetteville, AR: University of Arkansas: 164–175.

Moll, J.E. 1996. A guide for road closure and obliteration in the Forest Service. San Dimas, CA: U.S. Department of Agriculture, Forest Service, San Dimas Technology and Development Center. 49 p.

Morrison, G.M.; Revitt, D.M.; Ellis, J.B. 1995. The gully pot as a biochemical reactor. Water Science and Technology. 31: 229–236.

Mungur, A.S.; Shutes, R.B.E.; Revitt, D.M.; House, M.A. 1994. An assessment of metal removal from highway runoff by a natural wetland. In: Kadlec, R.H.; Brix, H., eds. Wetland systems for water pollution control 1994: Selected proceedings for the 4th international conference on wetland systems for water pollution control; 1994 November 6–10; Guangzhou, China. London: Middlesex University, Urban Pollution Research Center: 169–175.

Nessim, M.; Stephens, M. 1998. Quantitative risk-analysis model guides maintenance budgeting. Pipe Line & Gas Industry. 81(6): 33–38.

Packer, P.E.; Christensen, G.F. 1977. Guides for controlling sediment from secondary logging roads. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station; Missoula, MT: Northern Region. 41 p.

Patric, J.H. 1976. Soil erosion in the eastern forest. Journal of Forestry. 74: 671–677.

Pugh, A.L., IV; Norton, S.A.; Schauffler, M. [and others]. 1996. Interactions between peat and salt-contaminated runoff in Alton Bog, Maine, U.S.A. Journal Hydrology Amsterdam. 182(1–4): 83–104.

R**eid, L.M.; Dunne, T.** 1984. Sediment production from forest road surfaces. Water Resources Research. 20(11): 1753–1761.

Revesz, K.; Coplen, T.; Baedecker, M.J. [and others]. 1996. Investigation of methane production and consumption by use of stable isotopes. In: Isotopes in water resources management. Vienna, Austria: International Atomic Energy Agency: 381–387.

Rice, R.M.; Tilley, F.B.; Datzman, P.A. 1979. A watershed's response to logging and roads: south fork of Caspar Creek, California, 1967–1976. Res. Pap. PSW–146. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 12 p.

Robinson, E.G.; Mills, K.; Paul, J. [and others]. 1999. Storm impacts and landslides of 1996. For. Res. Tech. Rep. 4. Salem, OR: Oregon State Department of Forestry. 145 p.

Romero-Lozano, J. 1995. Highway design considers stormwater drainage and treatment options. Public Works. 126(1): 32–33.

Rummer, B.; Stokes, B.; Lockaby, G. 1997. Sedimentation associated with forest road surfacing in a bottomland hardwood ecosystem. Forest Ecology and Management. 90(2–3): 195–200.

Sansalone, J.J.; Buchberger, S.G. 1995. An infiltration device as a best management practice for immobilizing heavy metals in urban highway runoff. Water Science Tech. 32(1): 119–125.

Shanley, J.B. 1994. Effects of ion exchange on stream solute fluxes in a basin receiving highway deicing salts. Journal of Environmental Quality. 23(5): 977–986.

Sittig, M. 1978. Petroleum transportation and production: oil spill and pollution control. Park Ridge, NJ: Noyes Data Corporation. 360 p.

Sonett, M.L. 1999. Integrated surface management for pipeline construction: the mid-America pipeline company four corners project. In: Finch, D.M.; Whitney, J.C.; Kelly, J.F.; Loftin, S.R., eds. Proceedings of the workshop on Rio Grande ecosystems: linking land, water, and people toward a sustainable future for the middle Rio Grande Basin; 1998 June 2–5; Albuquerque, NM. Publ. RMRS–P–7. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 180–183.

Stalder, F. 1997. Pipeline failures. Materials Science Forum. 247: 139–146.

Startin, J.; Lansdown, R.V. 1994. Drainage from highways and other paved areas: methods of collection, disposal, and treatment. Journal of Institute of Water and Environmental Management. 8(5): 518–526.

Swift, L.W., Jr. 1984a. Gravel and grass surfacing reduces soil loss from mountain roads. Forest Science. 30(3): 657–670.

Swift, L.W., Jr. 1984b. Soil losses from roadbeds and cut and fill slopes in the Southern Appalachian Mountains. Southern Journal of Applied Forestry. 8(4): 209–215.

Swift, L.W., Jr. 1986. Filter strip widths for forest roads in the Southern Appalachians. Southern Journal of Applied Forestry. 10(1): 27–34.

Teal, M.J.; Carreon, S.; Bradley, J.B. 1995. Avoiding pipeline failures at waterway crossings. 2: Scour countermeasures. In: International conference on advances in underground pipeline engineering: Proceedings of the 2d international conference on advances in underground pipeline engineering; 1995 June 25–28; Bellvue, WA. New York: American Society of Civil Engineers: 77–78.

Thomson, N.R.; McBean, E.A.; Snodgrass, W.; Monstrenko, I.B. 1997. Highway stormwater runoff quality: development of surrogate parameter relationships. Water, Air and Soil Pollution. 94(3–4): 307–347.

U.S. Department of Agriculture. 1989. The 2d RCA appraisal. Soil, water, and related resources on non-Federal land in the United States: analysis of condition and trends. Washington, DC: U.S. Department of Agriculture. 42 p.

U.S. Department of Agriculture, Forest Service. 1990. R1–WATSED Region 1 water and sediment model. Missoula, MT: U.S. Department of Agriculture, Forest Service, Region 1. 120 p.

U.S. Department of Agriculture, Forest Service. 1998. Forest Service roads: a synthesis of scientific information. Washington, DC: U.S. Department of Agriculture, Forest Service. 52 p.

U.S. Department of the Interior, Bureau of Land Management. 1978. Draft environmental statement: crude oil transportation system, Port Angeles, WA, to Clearbrook, MN (as proposed by Northern Tier Pipeline Company). Washington, DC: U.S. Government Printing Office. 187 p.

U.S. Department of Transportion. 1999. Pipeline statistics: liquid accident yearly summaries. Washington, DC: U.S. Department of Transportation, Office of Pipeline Safety. [Available on the Web at: http://ops.dot.gov].

Vowell, J.L. 1985. Erosion rates and water quality impacts from a recently established forest road in Oklahoma's Ouachita Mountains. In: Blackmon, B.G., ed. Proceedings; forestry and water quality: a mid-South symposium; 1985 May 8–9; Little Rock, AR. Fayetteville, AR: University of Arkansas: 152–163.

Wan, M.T. 1994. Utility right-of-way contaminants: polycyclic aromatic hydrocarbons. Journal of Environmental Quality. 23(6): 1297–1304.

Wemple, B.C.; Jones, J.A.; Grant, G.E. 1996. Channel network extension by logging roads in two basins, western Cascades, Oregon. Water Resources Bulletin. 32(6): 1195–1207.

Wilson, B.R. 1996. Pipeline failure investigations: analytical techniques and case studies. In: Proceedings of the 1st annual international pipeline conference; 1996 June 9–13; Calgary, Canada. New York: American Society of Mechanical Engineers: 409–416.

Yaorong, F.; Helin, L.; Pingsheng, Z. [and others]. 1996. Failure analysis and fitness-for-service assessment for a pipeline. In: Process and power: Proceedings of the American Society of Mechanical Engineers pressure vessels and piping conference; 1996 July 21–26; Montreal, Canada. New York: American Society of Mechanical Engineers: 25–30.

Yoho, N.S. 1980. Forest management and sediment production in the South—a review. Southern Journal of Applied Forestry. 4(1): 27–36.

Part III:

Effects of Vegetation Management on Water Quality

Logs are kept wet at the woodyard, Mississippi. Photo by Bill Lea

Chapter 10

Timber Management

John D. Stednick[1]

Introduction

Forest management activities that disturb the soil or remove vegetation may potentially affect the quality of drinking water sources. Examples include removing trees from the site for timber harvest, forest stand regeneration, and stand improvement. Soil disturbance from tree felling is minor, but movement of logs or whole trees to a landing or collection point may disturb the soil surface. Other soil surface disturbances may be related to collection and haul roads. Roads are addressed separately in chapter 9. Stand improvement may include selective harvesting of trees in either dominant or suppressed crown positions. Forest stand thinning may increase water and nutrient availability, but any increase is utilized quickly by the remaining vegetation. Stand improvement may also include subordinate vegetation removal by fire (see chapter 12) or by herbicides (see chapter 13).

This chapter reviews the potential effects of timber management on water quality. Forest vegetation management may affect concentrations of suspended sediment and nutrients in surface water and stream temperature.

Erosion/Sedimentation

Forest management activities associated with timber harvesting may affect the physical, chemical, and biological properties of the soil. If these activities increase soil erosion, then water quality may be decreased through suspended sediment transport or stream sedimentation. Soil erosion is the detachment and movement of soil particles. It is measured as tons per acre per year [metric tonnes (Mg) per hectare per year]. Suspended sediment is eroded soil material transported in the water column of a stream. It is measured as a concentration such as milligrams per liter or as turbidity, which is an optical measurement of the water's ability to diffract light and is expressed as nephelometric turbidity units (Stednick 1991).

Site properties that affect erosion processes include vegetative cover, soil texture, soil moisture, and slope, among others (Falletti 1977, Renfro 1975). The sediment load of streams (both suspended and bed load) is determined by such characteristics of the drainage basin as geology, vegetation, precipitation, topography, and land use. Sediment enters the stream system through erosion processes. To achieve stream stability, an equilibrium must be sustained between sediment entering the stream and sediment transported through the channel. A land-use activity that significantly changes sediment load can upset this balance and result in physical and biological changes in the stream system (State of Idaho 1987).

The existing form and characteristics of streams have developed in a predictable manner as a result of the water and sediment load from upstream. Natural channels are self-formed and self-maintained. Both water and sediment yields may change due to timber management or other land-use activities upstream.

Issues and Risks

The forest practices with the greatest potential for causing erosion and stream sedimentation are road construction, tractor skidding of logs, and intensive site preparation. These activities can contribute to surface, gully, and large-mass soil movements (see chapters 3, 9). Other soil erosion processes may occur at smaller scales and rates. Generally, as site disturbance increases, soil erosion increases.

Most soil erosion studies only measure the amount of soil moved or displaced. The actual amount of eroded soil reaching the surface water is a small percent (2 to 10 percent) of the erosion occurring in the watershed. This percentage is termed the sediment delivery ratio and is the amount of sediment produced divided by the amount of soil erosion as a function of the watershed area (Dunne and Leopold 1979). Soil erosion and subsequent sediment delivery to the stream usually occurs at a specific location or locations downstream from the disturbance.

Sediment accumulation in stream channels may adversely affect water quality and aquatic life. Stream sedimentation

[1] Professor and Program Leader, Colorado State University, Watershed Science, Fort Collins, CO.

may adversely affect stream macroinvertebrates, intergravel dissolved oxygen, and intergravel flow and migration paths. When waters with increased sediment or turbidity are used for drinking water, treatment costs increase. The water must be filtered or stored to allow settling to remove suspended sediment. Often chlorination rates must be increased to disinfect water with elevated suspended sediment because bacteria may be associated with the sediment. See chapter 3 for further discussion of sediment effects.

Findings from Studies

Undisturbed forest watersheds usually have erosion rates from near 0 to 0.25 tons per acre per year (0.57 Mg per hectare per year) (Binkley and Brown 1993a). Erosion rates have been estimated as <0.1 tons per acre per year (0.2 Mg per hectare per year) for three-quarters of eastern and interior western forests (Patric and others 1984). Typical timber harvesting and road construction activities may increase erosion rates to 0.05 to 0.25 tons per acre per year (0.11 to 0.57 Mg per hectare per year) (table 10.1). More intensive site preparation treatments such as slash windrowing, stump shearing, or roller chopping may increase soil erosion rates by up to 5 tons per acre per year (11.4 Mg per hectare per year). Erosion from unpaved road and trail surfaces may be higher yet (see chapter 9).

Numerous studies have been done on the effects of different forest management practices on erosion rates or sediment production (table 10.1). In general, increased site disturbance will result in increased soil erosion and subsequent sediment production. The type and magnitude of erosion depend on the amount of soil exposed by management practices, the kind of soil, steepness of the slope, weather conditions, and any treatments after the disturbance (Swank and others 1989).

Logging in the Southeastern United States increased erosion to 1.8 tons per acre per year (4.1 Mg per hectare per year) from the undisturbed rate of 0.005 tons per acre per year (0.011 Mg per hectare per year); about 10 percent of the increase was attributed to site preparation (Hewlett 1979). Roller chopping and slash burning in North Carolina had little effect on soil erosion after harvest, but soil disking and herbicide application increased soil erosion to 4.5 tons per acre per year (10 Mg per hectare per year) (Pye and Vitousek 1985).

Timber harvesting and subsequent yarding can increase sediment in streams by increasing surface erosion rates and increasing the risk of mass soil movement (Brown and Krygier 1971, Brown and others 1976, Davis 1976). Site disturbance can reduce infiltration rates and increase overland runoff and related surface erosion.

Logs are moved (skidded) from the stump to a landing by tractor, cable, aerial systems, or animals. Tractor skidders may be either crawler or wheeled units, both of which are frequently equipped with arches for reducing the extent of contact between log and ground. Site disturbance will vary greatly with the type of skidding or yarding system. Crawler tractors generally cause the greatest amount of site disturbance, followed closely by wheeled skidders, but on some sites use of wheeled skidders can result in more compaction than crawler tractors (Bell and others 1974, Davis 1976). One method of decreasing the amount of soil disturbed by crawler tractors or wheeled skidders is through careful layout of skid trails (Rothwell 1971). Careful location of skidroads can greatly decrease the impact of tractor logging. Cable logging systems will result in less site disturbance because yarding trails are established to the yarding tower machinery, which is restricted to road surfaces. Cable systems can be ranked in order of decreasing soil disturbance as follows: single drum jammer, high lead cable, skyline, and balloon (Brown and others 1976, Davis 1976, Stone 1973). Helicopters and balloons will likely result in minimum site disturbance, but both are costly and subject to operational constraints.

Unlike many other land uses that disturb soil for long periods, any increase in sediment yields from timber management activities is usually short-lived. Surface soil disturbances provide a sediment supply, but once the finer materials are transported and as revegetation occurs, that site is less apt to continue eroding. Sediment yields or measured suspended sediment concentrations decrease over time as a negative exponential (Beschta 1978, Leaf 1974, Megahan 1975, NCASI 1999a). This time factor should be considered when assessing watersheds for impacts on drinking water (Stednick 1987). Swank discusses sediment yields over time as the forest succession after logging proceeds (see chapter 11).

Raindrop splash may potentially sort surface soil particles and create an armor layer or erosion pavement. Erosion pavements can form quickly on some soils in the West, discouraging further erosion. In the South, however, many surface soils have fine texture to depths of several inches (centimeters) to several feet (meters). There, the soil surface often becomes sealed, accelerating surface runoff, erosion, and sedimentation. Fine soil particles continue to be transported by surface runoff until the area is completely revegetated. Revegetation may take 2 years where trees have been harvested, 3 to 5 years for skid trails and temporary logging roads, and 3 to 5 years for site preparation depending on the type of practice.

Table 10.1—Effects of various timber harvests or site preparations on soil erosion and sediment production

Location	Treatment	Erosion or sedimentation rate	Unit	Measurement	Reference
East					
Georgia Piedmont	Control	0.002	Tons/ac/yr	Sediment	Hewlett 1979
	Harvest	1.8	Tons/ac/yr		
	Roading	1.6	Tons/ac/yr		
North Carolina	Roller-chop and burn	1.8	Tons/ac/yr	Erosion traps	Pye and Vitousek 1985
	Shear stumps, windrow slash	1.8	Tons/ac/yr		
	Above plus herbicide (glyphosate)	4.5	Tons/ac		
Southeast United States	Natural	0 – .2	Tons/ac/yr	Erosion	Burger 1983
	Harvest with roads	.05 – .23	Tons/ac/yr		
	Burn	.02 – .32	Tons/ac/yr		
	Chop	.02 – .11	Tons/ac/yr		
	Chop and burn	.07 – .18	Tons/ac/yr		
	Windrow slash	.09 – .11	Tons/ac/yr		
	Disk	1.13 – 4.5	Tons/ac/yr		
Coweeta Hydrologic Laboratory, NC	Poor road design	840	Yd^3/mi of road	Volume/road length	Swift 1988
	Poor road design	5700	mg/L	Sediment conc.	
	Good road design with grass or gravel	.02 – .3	Tons/ac		
Hubbard Brook Experimental Forest, NH	Natural (WS4-WS6)	.011	Tons/ac/yr	Erosion	Hornbeck and others 1987
	Harvest and herbicide	.05	Tons/ac/yr		
Fernow Experimental Forest, WV	Natural	.05	Tons/ac/yr	Erosion to stream	Aubertin and Patric 1972, 1974
	Harvest	.02	Tons/ac/yr		
Cherokee County, TX	Natural	.008	Tons/ac/yr	4-yr average	Blackburn and others 1986
	Harvest, chop, and burn	.006	Tons/ac/yr		Blackburn and Wood 1990
	Harvest, shear, windrow, and burn	.36	Tons/ac/yr		
Natchez, TN	Control	82	mg/L	Stormflow sediment concentration	McClurkin and others 1985
	Harvest only	183	mg/L		
Gulf coastal Mississippi	Control	0.2	Tons/ac/2 yr	Erosion over 2 yr	Beasley 1979
	Harvest, chop, and burn	6.7	Tons/ac/2 yr		
	Harvest, shear, and windrow	6.7	Tons/ac/2 yr		
	Harvest, shear, and windrow and plow beds	9.0	Tons/ac/2 yr		

continued

Table 10.1—Effects of various timber harvests or site preparations on soil erosion and sediment production (continued)

Location	Treatment	Erosion or sedimentation rate	Unit	Measurement	Reference
East (continued)					
South Carolina	Clearcut	.07	Tons/ac/yr	Erosion	Van Lear and others 1985
	Control	.01	Tons/ac/yr		
Sumter National Forest, SC	Low-intensity burn	.06	Tons/ac/yr	Erosion	Robichand and Waldrop 1994
	High-intensity burn	2.6	Tons/ac/yr		
West					
Oregon	Roads	> .25	In./yr	Road surface	Fredriksen 1965
Fraser Experimental Forest, CO	Control	.018	Tons/ac/yr	To stream	Leaf 1974
	Roads and harvest—Fool Creek	.005	Tons/ac/yr		Leaf 1974
	Roads and harvest—Deadhorse	.014	Tons/ac/yr		Stottlemeyer 1987
H.J. Andrews Experimental Forest, OR	Control WS9	.014	Tons/ac/yr	To stream	Sollins and others 1980
	Clearcut WS10	.09	Tons/ac/yr		
Ouachita Mountains, AR	Control	.005	Tons/ac/yr	Surface erosion	Miller and others 1988
	Harvest, roller-chop, and burn	.09	Tons/ac/yr		
	Harvest, selection cut	.018	Tons/ac/yr		
Silver Creek, ID	Clearcut with buffer	5.8	Tons/ac/yr—1st yr	Erosion	Clayton and Kennedy 1985
		1.8	Tons/ac/yr—2d yr		
Beaver Creek, AZ	Control	.009 – .09	Tons/ac/yr	Annual range	Ward and Baker 1984
	Control	11	Tons/ac/yr	Observed maximum	
	31% clearcut	1.3	Tons/ac/yr	Observed maximum	
	100% clearcut	27.4	Tons/ac/yr	Observed maximum	
	Clearcut	.14	Tons/ac/yr	Plot erosion	Heede and King 1990
Caspar Creek, CA	Selective clearcut	4.5	Yd^3/ac/yr	Erosion	Krammes and Burns 1973
Drew County, AR	Clearcut	.120	Tons/ac/yr	Erosion	Beasley and Granillo 1988
	Selection cut	.005	Tons/ac/yr		
	Control	.002	Tons/ac/yr		
Clark County, AR	Clearcut—mechanical prep	.24	Tons/ac/yr	Erosion	Beasley and others 1986
	Clearcut—chemical prep	.11	Tons/ac/yr		
	Control	.03	Tons/ac/yr		

Conc. = concentration, WS = watershed.

Some form of site preparation is often needed to ensure the establishment of tree reproduction after timber harvest. The purpose of site preparation is to provide the environmental conditions necessary for seed or seedling survival and early growth. Site preparation usually involves providing a mineral seedbed and controlling competing and non-desirable vegetation. Site preparation treatments include fire, herbicide application, slashing and windrowing, roller chopping, soil disking, or other mechanical techniques. Fertilizer may be applied to help establish seedlings and to speed their growth after establishment.

In the Southeastern United States, upland hardwood stands are sometimes converted to pine (*Pinus* spp.). Site preparation treatments include burning or chemical treatments to kill the existing vegetation. Soils in the region are often fine textured and deep and may continue to erode at an accelerated rate for a few years. A winter burn and herbicide application increased stormflows, overland flows, peakflows, and sediment production from two small watersheds in northern Mississippi (Ursic 1970). Three years after the fire, when monitoring ended, most of the hydrologic effects were still evident.

Suspended sediment transport varies with the areal extent of the soil disturbance, nearness of a stream, and stream energy. Suspended sediments are often fine-textured materials with large surface areas per unit of weight. These large surface areas are reactive and may adsorb and absorb various constituents including phosphorus, introduced chemicals, and petroleum products.

Streamside vegetation or filter strips have been used to prevent overland flow and soil erosion from reaching surface waters. The filter strip, or equivalent, decreases the velocity of the overland flow by surface roughness. The decreased velocity allows sediment to settle out and overland waters to infiltrate into the undisturbed soils. The streamside vegetation filters were originally used to control or limit road-derived sediment from reaching forest streams. The filter was a recommended width and was dependent on hillslope. These filter strips are effective in sediment removal unless an extreme precipitation or overland flow event exceeds the sediment detention/retention capacity. The characteristics that make filter strips work include width, vegetative and litter cover, surface roughness, and microtopography. Microtopography allows overland flow to concentrate in certain areas and flowpaths. Control of road-derived sediment migration is frequently by these strips. The effectiveness of filter strips on controlling soil erosion for most harvest and site preparation practices has not been rigorously tested.

Routing and storage are particularly important components in the transport of sediment through the stream system. They are critical to the quantification of short- and long-term impacts of land-use activities on the quality of drinking water sources. However, the storage and routing processes are highly variable and do not exhibit steady-state behavior (see chapter 3).

Catchment studies have identified correlations between annual peak discharge and annual sediment discharge and between total annual flow and annual sediment discharge (NCASI 1999a). Altering flow and erosion may upset channel stability, increasing turbidity and sediment concentrations to drinking water sources.

Reliability and Limitations of Findings

Studies have shown that increased site disturbance has the potential to increase soil erosion and sediment production. Soil erosion and sediment yield from undisturbed forest watersheds are low. Site disturbance from timber harvesting activities vary by logging and yarding techniques, site preparation practice, operator techniques, soil vegetative cover, slope, soil moisture, soil depth, and soil texture among other environmental factors. Soil erosion processes are well understood, and models have been developed for regional predictions of soil erosion throughout the United States.

Measuring instantaneous sediment concentration (and turbidity) in small streams is relatively easy. Measuring soil erosion is not. Erosion is variable in time and space, and the eroded soil must reach the stream channel to become sediment. Once in the stream channel, most of the sediment is transported irregularly when streamflows are high. Sediments may be stored in the channel and released over a long period. In-channel disturbances may create in-channel sediment sources, separate from the hillslope processes. Large sediment inputs to stream channels can be assessed by monitoring the physical features of the channel (MacDonald and others 1991, State of Idaho 1987). Such features include channel width-to-depth ratios, pool volume occupied by sediment, and substrate size and particle size distribution.

Research Needs

1. There is no standard or protocol for erosion plot research on forest land. A standard research method for soil erosion studies should be decided upon.
2. The importance of dry ravel as an agent of erosion needs further investigation.

3. Research is needed on routing eroded soil to streams. Erosion does not equal suspended sediment. Measured erosion rates do not or should not imply that eroded soil is reaching the stream channel. Suspended sediment monitoring is not difficult, but requires labor and equipment that may not be available. For source areas the question is: Do suspended sediment concentrations best measure the effects of site disturbance?

4. Recommendation and design of vegetative filter strips are often based on width only. Research needs to better define the characteristics that control sediment movement including slope, vegetative and litter cover, runoff velocity and volume, surface roughness, and microtopography of the filter strip and disturbed area above.

5. Research is needed on monitoring of stream channel geomorphologic features, which may provide a good measure of land-use effects, particularly multiple or cumulative effects. Increased annual water yield from timber harvesting has been well documented, but the effect of timber harvesting on peak flows is less clear. Can this altered hydrology increase sediment transport from in-channel sources and result in changes in channel morphology? Conversely, how much increased sediment input can a stream segment receive without changes in channel morphology?

Key Points

Site disturbance may result in soil compaction and decreased infiltration capacity. If infiltration capacity is exceeded by precipitation intensity, overland flow may result in soil erosion and suspended sediment production. Even undisturbed forest watersheds produce sediment, mostly from in-channel sources. Sediment impacts from timber management activities can be minimized by:

1. Careful planning, supervising, and implementing of forest practices.

2. Keeping the treatment area small and hydrologically isolated.

3. Leaving adequate filter strips between treatment areas and streams.

4. Maintaining ground cover in the treatment area to reduce surface runoff and erosion, and increasing the effectiveness of filter strips to trap eroded soil before it enters the stream.

5. Operating during the season with the lowest erosion risk.

Stream Temperature

Issues and Risks

Forest management activities can increase, maintain, or decrease water temperature. Such changes can affect drinking water quality (chapter 2) by altering dissolved oxygen and survival rates of pathogens.

Findings from Studies

Surprisingly few studies have been published on the effects of silvicultural practices on water temperature, and most of these were conducted in the 1970's (table 10.2). These studies include harvesting with and without streamside vegetation buffers. Several synthesis papers indicate that few additional temperature studies have been conducted (Beschta and others 1987, Binkley and Brown 1993a, Swank and Johnson 1994).

Exposure of small streams to direct solar radiation is the dominant process for stream temperature increases (Tiedemann and others 1988). Other mechanisms including increased air temperature, channel widening, soil water temperature increases, and streamflow modification have been proposed [Ice, in press (a)]. Small streams with smaller surface areas may be more susceptible to heating, but usually return to expected temperature within 500 feet [150 meters (m)] downstream [Andrus and Froehlich 1991; Ice, in press (b)]. Maintaining shade in riparian zones can be used to avoid most temperature increases in small streams. As stream width increases, more of the water surface is exposed to sunlight and the influence of riparian canopy on stream temperature decreases.

Literature on the effects of timber harvesting on stream temperatures (table 10.2) shows daily maximum stream temperature increases from 1.2 to 7.2 °C in eastern forests and 0.6 to 8 °C in western forests. The range in temperature increases reflects a range in streamside vegetation buffers from no buffer to a 100-m buffer. Changes in minimum nighttime stream temperatures (during the winter or dormant season) range from no change to < 1 °C in the East and from zero to < 2 °C in the West.

Reliability and Limitations of Findings

Stream temperatures in small streams may increase after timber harvesting when the streamside vegetation canopy is removed. This effect can be mitigated by maintaining streamside buffers. Several studies have reported temperature increases with streamside buffers, but increases are much smaller than for fully exposed streams. The lack of

Table 10.2—Effects of timber harvesting with and without streamside buffers on stream temperature

Location	Treatment	Maximum temperature: Temperature	Change	Measure	Reference
		- - - - Degree Celsius - - - -			
East					
Georgia	Clearcut with buffer	25.0	3.9	Average daily	Hewlett and Fortson 1982
	Control	21.1			
Maryland	Riparian harvest		4.4 –7.6	Summer max.	Corbett and Spencer 1975
Coweeta Hydrologic Laboratory, NC	100% clearcut with no buffer	21.7	3.4	Average daily	Swift and Messer 1971
	Control	18.3			
Newark, NJ	Riparian herbicide		3.3	Avg. summer max.	Corbett and Heilman 1975
Fernow Experimental Forest, WV	95% clearcut with buffer removed	16.1	1.7	Average weekly	Aubertin and Patric 1974
	Control	14.4			
	Plot harvest		4.0	Summer max.	Kochendorfer and Aubertin 1975
Hubbard Brook Experimental Forest, NH	100% clearcut with no buffer	20.0	4.0	Average daily	Likens and others 1970
	Control	16.0			
Pennsylvania State Forest, PA	Riparian harvest		3.9	Summer max.	Lynch and others 1975
Leading Ridge, PA	Control	19.4	1.2	Average daily	Rishel and others 1982
	44% clearcut with buffer	20.6			
	Control	17.8	7.2	Average daily	Rishel and others 1982
	85% clearcut with no buffer	25.0			
West					
Alsea, OR	Control	12.2	10.0	Average daily	Brown and Krygier 1970
	85% clearcut with no buffer	22.2	16.0	Summer max.	
Steamboat, OR	Control	14.4	.6	Daily max.	Brown and others 1971
	Clearcut with buffer	15.0			
	Control	13.3	2.3	Daily max.	Brown and others 1971
	Clearcut with no buffer	15.6			
British Columbia	Clearcut		.5–1.8	Average daily	Holtby and Newcombe 1982
H.J. Andrews Experimental Forest, OR	Clearcut		4.4–6.7	Daily max.	Levno and Rothacher 1969
Coyote Creek, OR	Clearcut		8.0	Daily max.	Harr and others 1979

documentation on buffer characteristics makes extrapolation difficult. Different measurements of stream temperature also make direct comparisons difficult. Studies have reported daily, monthly, or seasonal maxima or mean temperatures. Within-stream temperature variability often is not considered in monitoring programs.

Attributes needed to estimate the contribution of forest overstory to stream surface shade include stream width, distance from vegetation to stream, stream orientation, height and density of vegetation, crown or canopy measurement, latitude, date, and time (Quigley 1981).

A simpler model developed to predict the effect of clearcutting on temperatures of small streams uses the calculated heat load to the stream surface area (Brown 1970). This or similar models should be validated before use. It would be difficult to suggest one streamwater buffer model as suitable for all forest watersheds, but measurement of the angular canopy density can determine the importance of a buffer strip to prevent stream temperature increases after timber harvesting. Angular canopy density is the projection of the streamside vegetation canopy measured at the angle above the horizon at which direct-beam solar radiation passes through the canopy (Beschta and others 1987).

Generally, forest practices that open small stream channels to direct solar radiation are the practices that increase stream temperatures. Retention of streamside vegetation appears to mitigate potential temperature changes, especially the greater temperature changes. These principles are well documented by research throughout the country. Streamside canopy removal may also decrease winter streamwater temperatures, since radiation losses may be increased. For small streams, temperature returns to undisturbed levels within a short distance downstream of where canopy shade is reestablished.

Accurate stream temperature assessments vary from a single instantaneous measurement to continuous measurement, depending on the stream diel and seasonal variations. Stream temperature data need to be evaluated over the long term. Statistical methods include harmonic analysis, time series, and trend analysis[2] (Hostetler 1991, Limerinos 1978).

[2] Stednick, J.D. 1999. Stream temperature trends in the New Alsea watershed study. [55 p.]. Unpublished report. On file with: Department of Earth Resources, Colorado State University, Fort Collins, CO 80523–1482.

Research Needs

1. Stream temperature monitoring and reporting protocols need to be developed.
2. The range or daily variation in temperature may increase after removal of streamside vegetation. Research is needed on these variations because they might affect drinking water quality.
3. Timber harvesting with proper streamside vegetation buffers should cause minimal stream temperature changes. Stream buffers are defined by width only. More studies need to be conducted investigating the efficiency of different components of streamside canopy cover on stream temperatures.
4. Stream temperature monitoring has tended to emphasize physical measurements of temperature. Remote sensing of stream temperature may provide more data on temperature changes over time and space.
5. Few water-quality related studies have assessed cumulative watershed effects. Temperature measurement studies at different spatial scales need to be conducted. Long-term temperature data are needed to place the potential effects of changes in stream temperature in the context of global or regional cycles of climate change or variability. Long-term records of stream temperature in undisturbed, forested watersheds need to be collected.

Key Points

In general, removal of streamside vegetation cover has the potential to increase streamwater temperatures during the day in the summer. In certain settings, the vegetation removal may allow for decreased nighttime temperatures, especially in the winter. Temperature changes return to pretreatment levels as the streamside vegetation reestablishes. Streamside vegetation to maintain a thermal cover over the stream is key to maintaining stream temperatures at existing levels.

Nutrients

Water from forested watersheds is typically lower in nutrients than water that drains from other lands. Forest management activities such as forest cutting and harvesting may increase annual water yields (Bosch and Hewlett 1982, Stednick 1996), interrupt the natural cycling of nutrients, and increase nutrient concentrations in streamwaters. Nitrogen and phosphorus cycles and their impacts on drinking water quality are discussed in chapter 2.

Issues and Risks

Forest management activities, such as timber harvest and fertilization, can increase nutrient concentrations in streams.

Findings from Studies

Nitrate nitrogen (NO3-N) concentrations are usually quite low (0.002 to 1.0 milligrams per liter) in streams draining undisturbed forest watersheds (Binkley and Brown 1993b). Concentrations are low because nitrogen is used rapidly by ecosystem biota and because nitrate formation (nitrification) is relatively slow in forest soils. Slow rates of organic matter decomposition, acid soil conditions common in forest environments, and bacterial allelopathy all decrease rates of nitrification. Organic matter and anaerobic conditions in saturated riparian soils allow for denitrification, which is the reduction of nitrate to nitrogen gas, which may be lost to the atmosphere.

Throughout the United States, studies in many areas have found that nutrient losses from silvicultural activities to be minimal and water quality not degraded (Aubertin and Patric 1974, Chamberlain and others 1991, Harr and Fredriksen 1988, Hornbeck and Federer 1975, Martin and others 1984, McClurkin and others 1987, Pierce and others 1972, Rense and others 1997, Sopper 1975, Swank 1988).

Nutrients contained in the organic matter in trees, litter, and soils can be affected by various forest management practices. Cutting vegetation disrupts the processes that regulate the nutrient cycle and may accelerate dissolved nutrient leaching and loss via streamflow. Exposing sites to direct sunlight may increase the rate of nitrogen mineralization. Nutrients associated with eroded soil particles and sediment may be lost from the site (Swank 1988). There is usually minimal opportunity for a buildup of these nutrients in the stream system after a timber harvest because of the normally brief period of increased nutrient flux to the stream (Currier 1980). Other nutrients rarely cause water-quality problems, and this discussion is limited to nitrogen and phosphorus.

Forest management activities such as harvesting or thinning may interrupt nutrient cycles, and nutrients may be released (Swank and Johnson 1994). Catchment studies have produced a large body of information on streamwater nutrient responses, particularly from clearcutting (table 10.3). Changes in streamwater nutrient concentrations vary substantially among localities, even within a physiographic region. In central and Southern Appalachian forests, nitrate-nitrogen, potassium (K+), and other constituents increased after harvesting, but the changes were small and did not affect downstream uses (Swank and others 1989).

Clear-cutting in northern hardwood forests may result in large increases in concentrations of some nutrients (Hornbeck and others 1987). Research on catchments has identified some of the reasons for varied ecosystem response to disturbance (Swank and Johnson 1994). Swank discusses the long-term nitrate-nitrogen trends after harvest in chapter 11. In areas that are experiencing nitrogen saturation from deposition of nitrogen compounds in air pollution, disturbances such as forest harvesting can produce increased nitrate levels in streams and ground water (Fenn and others 1998). See chapter 3 for discussion of nitrogen-saturation effects.

Soil development factors and forest management strategies influence the rate of nutrient exports after timber harvesting (Swank and Johnson 1994). The rotation length, the time interval between timber harvests, is critical in determining the sustainability of harvest. Nutrient loss by leaching to streams is usually minor compared to the nutrient loss by biomass removal (Clayton and Kennedy 1985, Federer and others 1989, Johnson and others 1988, Mann and others 1988, Martin and Harr 1989). Nutrient loss differences are also observed between whole tree, saw log, or bole-only harvesting.

Phosphorus (P) occurs in several forms in surface water including the dissolved forms of orthophosphates and dissolved complex organics and in particulate forms (organic and inorganic) [Ice, in press (b)]. Phosphorus sources come from dry deposition (dust), wet deposition, and geologic weathering. Geology is a key factor in phosphorus concentrations from forests. Forest watersheds with more easily weathered rock, such as sedimentary or volcanic tuff and breccia, have higher instream concentrations than watershed with resistant rock, such as intrusive igneous. Dissolved phosphorus is probably one of the least responsive water-quality constituents to forest management.

Total phosphorus is strongly associated with soil particles or suspended sediment. Practices that increase or reduce sediment have similar effects on total phosphorus [Ice, in press (b)].

In general, nutrient mobility from disturbed forests follows the order: nitrogen > potassium > calcium and magnesium > phosphorus. Thus, forest harvesting or other disturbances, such as fire, will generally produce larger differences in nitrogen concentrations in streamwater than other constituents. Possible exceptions are the loss of calcium and potassium documented in the Northeastern United States where precipitation inputs had greater acidity from fossil fuel combustion (Federer and others 1989).

Table 10.3—Effects of clearcutting with and without buffers on mean annual nitrate-nitrogen, ammonium-nitrogen, and total-phosphorus concentrations

Location	Treatment	Mean concentration NO_3-N	NH_4-N	Total P[a]	Reference
		- - - *Milligrams per liter* - - -			
East					
Marcell Experimental Forest, MN	74% clearcut	0.16	0.55		Verry 1972
	Control	.12	.41		
Hubbard Brook Experimental Forest, NH					
WS2	100% cut and herbicide	8.67 – 11.94	.04 – .05	0.002	Likens and others 1970
WS4	33% strip cut	.19 – .20			
WS6	Control	.16 – .29	.05 – .09	.001	
White Mountain, NH					
Seven catchments	Control	.02 – .81			Pierce and others 1972
	Clearcut	1.31 – 3.84		.01 – .02	
Upper Mill Brook	Control	.23 – .27		.02 – .03	Stuart and Dunshie 1976
	Clearcut	.23 – .96			
Leading Ridge, PA					
LR2	100% clearcut and herbicide	.10 – 8.4			Corbett and others 1975
LR1	Control	.02 – .04			
Fernow Experimental Forest, WV					
WS3	100% clearcut	.18 – .49	.14 – .35	.04 – .07	Aubertin and Patric 1972, 1974
WS4	Control	.10 – .32	.13 – .48	.02 – .04	
Coweeta Hydrologic Laboratory, NC					
WS2	Control	.004	.002	.006	Douglass and Swank 1975
WS28	100% clearcut	.094	.003	.004	
West					
H.J. Andrews Experimental Forest, OR	Control	.020 – .200		.016 – .032	Fredriksen and others 1975
	100% clearcut	.001 – .010		.024 – .039	
Bull Run, OR	25% clearcut	.002 – .093	.001 – .005	.011 – .032	Fredriksen 1971
	Control	.002 – .013	.002 – .005	.014 – .040	
Coyote Creek, OR	100% clearcut	.001 – .275	.001 – .018	.062 – .100	Harr and others 1979
	Control	.001 – .005	.001 – .014	.036 – .060	Adams and Stack 1989
Chicken Creek, UT	13% clearcut	.025			Johnston 1984
	Control	.008			
Alsea, OR	85% clearcut	.19 – .44			Brown and others 1973
	Control	1.18 – 1.21			
Priest River, ID	Control	.20			Snyder and others 1975
	100% clearcut	.18			
Fraser Experimental Forest, CO	33% clearcut	.06			Stottlemeyer 1987
	Control	.006			
Beaver Creek, AZ	Control	.010			Ryan as cited by Binkley and Brown 1993b
	Clearcut	.220			

LR = Leading Ridge; NO_3-N = nitrate-nitrogen; NH_4-N = ammonium-nitrogen; total P = total phosphorus; WS = watershed.
[a] Blank columns represent no data collected.

Reliability and Limitations of Findings

Research has documented that timber harvesting may increase nitrate concentrations in soil water and streams. This finding is generally accepted without controversy. Soluble phosphorus concentrations are essentially unaffected by timber harvesting activities. Total phosphorus concentrations are closely linked to sediment concentrations. Some forest types in the United States have few studies investigating the influence of forest practices on water quality. The rather consistent streamwater chemistry response to timber harvesting allows response extrapolation.

However, an often erroneously cited study as an example of timber harvesting effects on water quality is an early Hubbard Brook study (Likens and others 1970). In this study, vegetation was cut, left onsite, and sprayed with a general herbicide for 3 years to kill any plant regeneration to research nutrient cycling processes. Nutrient concentrations, particularly nitrate, increased significantly. This watershed treatment was not representative of timber harvest and does not represent the effects of a typical timber harvest on water quality.

If vegetation is quickly reestablished, nutrient exports are short-lived and usually do not represent a threat to water quality or site productivity. There are a couple of possible exceptions. Nitrogen deposition can accumulate in forest soils over time, especially in areas with air-quality concerns (Riggan and others 1985, Silsbee and Larson 1982). If timber harvesting occurs in these areas, mobilization of accumulated soil nitrogen may result in higher nitrate concentrations and outputs in the streamwater (see chapter 3).

In the Pacific Northwest, water-quality samples from streams in forests with nitrogen fixing alder (*Alnus* spp.) may have higher nitrate concentrations than streams without alder (Binkley and Brown 1993b, Miller and Newton 1983). Since nitrogen is being added to the site by fixation, losses in site productivity are not a concern, but nitrate concentrations may be high enough to affect downstream uses.

Forest harvesting practices that minimize site disturbance and quickly establish new stands seem to minimize any potential water-quality effects. Streamside vegetation buffers are effective for sediment removal and nutrient removal.

Research Need

Soil water usually has higher nutrient concentrations than surface or streamwater. Changes in water chemistry at large scales (watershed to landscape) need to be evaluated, especially in the context of multiple land-use activities in time and space for cumulative watershed effects.

Key Point

Timber harvesting may increase nutrient concentrations in streams, especially nitrate, but any increase is usually short-lived. Watershed studies show that nutrient concentrations in soil water may be higher than concentrations in surface water suggesting that other water dilutes off-site concentrations.

Fertilizer

Urea fertilizer is highly soluble in water and readily moves into the forest floor and soil with any appreciable amount of precipitation. Under normal conditions, urea is rapidly hydrolyzed (4 to 7 days) to the ammonium ion (NH_4-N). When moisture is limited, urea may be slowly hydrolyzed on the forest floor. Rather than moving into the soil as ammonium, the increased soil surface pH favors formation of ammonia (NH_3-N), which is lost by volatilization. Volatilization losses may be significant. Fertilizer usually is applied in the spring or fall to take advantage of seasonal precipitation.

Fertilizers may enter surface water by several routes. Direct application of chemicals to exposed surface water is the most significant. Identification of surface water bodies prior to the application essentially eliminates this entry mode. When fertilizers are volatilized, ammonia absorption by surface water is minimal (U.S. Department of Agriculture, Forest Service 1980).

Issues and Risks

The issues and risks associated with fertilizer application are essentially the same as described in the Nutrients section, except if inadvertently applied to streams.

Findings from Studies

The reported effects of forest fertilization on water quality, particularly nutrient concentrations in streams, are variable[3] (reviews by Binkley and Brown 1993b, Binkley and others 1999, Bisson and others 1992, Fredriksen and others 1975). Nutrient retention by forest soils is excellent. Nutrient

[3] Stephens, R. 1975. Effects of forest fertilization in small streams on the Olympic National Forest, fall 1975. Unpublished report. 40 p. On file with: USDA Forest Service, Olympia Forestry Sciences Laboratory, 3625 93rd Avenue, South, Olympia, WA 98512 .

concentrations in surface waters after forest fertilization are usually low (table 10.4). Exceptions may occur in areas experiencing nitrogen saturation from air pollution inputs. For example, Fernow Experimental Forest, WV, a site that shows signs of nitrogen saturation (Fenn and others 1998), experienced high streamwater nitrate response to nitrogen fertilization (table 10.4). Ammonium-nitrogen and phosphorus are very reactive with forest soils and are retained on site. Ammonium-nitrogen concentration may increase in surface water as a result of direct fertilizer application to open water. Ammonium-nitrogen concentrations, however, are rapidly reduced through aquatic organism uptake and stream sediment sorption. See chapter 3 for discussion of surface and ground water responses to nitrogen additions in nitrogen-saturated watersheds.

Nitrate-nitrogen concentrations measured in surface water usually peak 2 to 4 days after fertilizer application (U.S Department of Agriculture, Forest Service 1980). The magnitude of the peak concentration may depend on the presence and width of streamside buffers and the density of small feeders and tributaries to the streams. Peak nitrate-nitrogen concentrations usually decrease rapidly but may remain above pretreatment levels for 6 to 8 weeks. Winter storms may also result in peak nitrate-nitrogen concentrations, but these peaks usually decrease over successive storms, and concentrations decrease quickly between storms.

Table 10.4—Effects of forest fertilization on maximum streamwater ammonium-nitrogen and nitrate-nitrogen concentrations

Location	Treatment	NH_4-N	NO_3-N	Reference
	Lbs/ac	*- - - Milligrams per liter*	*- - - -*	
East				
Fernow Experimental Forest, WV	230	0.8	19.8	Aubertin and others 1973
West				
Coyote Creek, OR	200	.04	.17	Fredriksen and others 1975
Olympic National Forest, WA	200	.02– .55	.07–3.85	Stephens 1975[a]
	200	.04	.121	Moore 1975
Entiat Experimental Forest, WA	48	<.02	.210	Klock 1971
	50		.068	Tiedemann and Klock 1973
Mitkof Island, AK	187	.003	2.36	Meehan and others 1975
Siuslaw River, OR	200	.49	7.6	Burrough and Froehlich 1972
Cascade Mountains, OR	200	<.01	<.25	Malueg and others 1972
Lake Chelan, WA	70	.011	.510	Tiedemann 1973
South Umpqua River, OR	200	.048	.177	Moore 1971
Ludwig Creek, WA	178	.004	2.7	Bisson and others 1992

NH_4-N = ammonium nitrogen; NO_3-N = nitrate nitrogen.
[a] Stephens, R. 1975. Effects of forest fertilization in small streams on the Olympic National Forest, fall 1975. 40 p. Unpublished report. On file with: Olympic National Forest, 1835 Black Lake Boulevard, SW, Olympia, WA 98512.

Reliability and Limitations of Findings

Relatively few studies have been published on the effect of forest fertilization on water quality, but results generally are consistent and suggest that concentrations of ammonium-nitrogen and phosphorus do not increase after fertilization (NCASI 1999b). Nitrate-nitrogen concentrations may increase, but increases are short-lived. Publications reviewed here suggest minimal water-quality changes under most conditions and appear universally applicable. Streamwater responses to fertilizer application are well understood and may be extrapolated. An exception to this generalization may be areas showing signs of nitrogen saturation. Nitrogen fertilization in these areas may increase stream nitrate.

Forest fertilization may increase nitrate-nitrogen concentrations by direct application of fertilizer to the stream or by a runoff-generating precipitation event after application. Careful delineation of application areas will avoid direct stream inputs. Fertilizer application timing with respect to seasonal precipitation or storm events minimizes fertilizer affects on water quality.

Research Needs

1. Streamside vegetation buffers or management zones are usually prescribed as a width. We need to know what specific components or processes in these streamside areas would minimize the movement of fertilizers into surface water.
2. Recent research identified certain bedrock materials as significant sources of nitrogen. Heretofore, geologic materials were not considered significant sources of nitrogen. How common are these materials?
3. What are the effects of repeated fertilizer applications in short-rotation forest plantations on water quality?
4. Response of stream nitrate to fertilization in areas experiencing nitrogen saturation is poorly understood and needs more study.

Key Points

Application of nitrogen or phosphorus fertilizers will not adversely affect surface waters including drinking waters, when the fertilizer is applied at a rate and time when the vegetation can use it. Fertilizer application should be timed to avoid rainy periods if fertilizer might be moved directly to surface waters. Streamside vegetation is effective in nutrient removal. Any increase in nutrient concentrations from fertilizer applications is usually short-lived and should not affect downstream uses.

Literature Cited

Adams, P.W.; Stack, W.R. 1989. Streamwater quality after logging in southwest Oregon. Proj. Comp. Rep. Suppl. PNW 87–400. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 28 p.

Andrus, C.W.; Froehlich, H.A. 1991. Riparian forest development after logging or fire in the Oregon coast range: wildlife habitat and timber value. In: The New Alsea watershed study. Tech. Bull. 602. New York: National Council for Air and Stream Improvement. 70 p.

Aubertin, G.M.; Patric, J.H. 1972. Quality water from clearcut forest land? Northern Logger and Timber Processor. March: 14–15, 22–23.

Aubertin, G.M.; Patric, J.H. 1974. Water quality after clearcutting a small watershed in West Virginia. Journal of Environmental Quality. 3: 243–249.

Aubertin, G.M.; Smith, D.W.; Patric, J.H. 1973. Quantity and quality of streamflow after urea fertilization on a forested watershed: first-year results. In: Forest fertilization symposium proceedings; [Date of meeting unknown]; [Location of meeting unknown]. Gen. Tech. Rep. NE–3. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 86–100.

Beasley, R.S. 1979. Intensive site preparation and sediment losses on steep watersheds in the Gulf Coastal Plain. Soil Science Society of America Journal. 43: 412–417.

Beasley, R.S.; Granillo, A.B. 1988. Sediment and water yields from managed forests on flat Coastal Plain sites. Water Resources Bulletin. 24(2): 361–366.

Beasley, R.S.; Granillo, A.B.; Zillmer, V. 1986. Sediment losses from forest management: mechanical vs. chemical site preparation after clearcutting. Journal of Environmenal Quality. 15(9): 413–416.

Bell, M.A.; Beckett, J.M.; Hubbard, W.F. 1974. Impact of harvesting on forest environments and resources. Ottawa, Ontario, Canada: Canadian Forest Service, Environment Canada. 39 p.

Beschta, R.L.; Bilby, R.E.; Brown, G.W. [and others]. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. In: Salo, E.O.; Cundy, T.W., eds. Streamside management: forestry and fishery interactions. Seattle: University of Washington: 191–232.

Beschta, Robert L. 1978. Long term patterns of sediment production following road construction and logging in the Oregon coast range. Water Resources Research. 14(6): 1011–1016.

Binkley, D.; Brown, T. 1993a. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin. 29(5): 729–740.

Binkley, D.; Brown, T. 1993b. Management impacts on water quality of forests and rangelands. Gen Tech. Rep. RM–239. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 114 p.

Binkley, D.; Burnham, D.H.; Allen, H.L. 1999. Water quality impacts of forest fertilization with nitrogen and phosphorus. Forest Ecology and Management. 121: 191–213.

Bisson, P.A.; Quinn, T.P.; Reeves, G.H.; Gregory, S.V. 1992. Best management practices, cumulative effects, and long-term trends in fish abundance in Pacific Northwest river systems. In: Naiman, R., ed. Watershed management: balancing sustainability and environmental change. New York: Springer-Verlag: 189–232.

Blackburn, W.H.; Wood, J.C. 1990. Nutrient export in stormflow forest harvesting and site-preparation in east Texas. Journal of Environmental Quality. 19: 402–408.

Blackburn, W.H.; Wood, J.C.; DeHaven, M.G. 1986. Stormflow and sediment losses from site-prepared forest land in east Texas. Water Resources Research. 22: 776–784.

Bosch, J.M.; Hewlett, J.D. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. Journal of Hydrology. 55: 3–23.

Brown, G.; Carlson, D.; Carter, G. [and others]. 1976. Meeting water quality objectives on State and private lands through the Oregon Forest Practices Act. Corvallis, OR: Oregon State University, Department of Forestry. 13 p.

Brown, G.W. 1970. Predicting the effect of clearcutting on stream temperature. Journal of Soil and Water Conservation. 28(1): 11–13.

Brown, G.W.; Gahler, A.R.; Marston, R.B. 1973. Nutrient losses after cleacut logging and slash burning in the Oregon coast range. Water Resources Research. 9: 1450–1453.

Brown, G.W.; Krygier, J.T. 1970. Effects of clearcutting on stream temperatures. Water Resources Research. 6: 1133–1140.

Brown, G.W.; Krygier, J.T. 1971. Clear cut logging and sediment production in the Oregon coast range. Water Resources Research. 7(5): 1189–1198.

Brown, G.W.; Swank, G.W.; Rothacher, J. 1971. Water temperature in the steamboat drainage. Res. Pap. PNW–119. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest Experiment Station. 17 p.

Burger, J.A. 1983. Physical impacts of harvesting and site preparation on soil. In: Maintaining forest site productivity: Proceedings of the 1st regional technical conference; [Location of meeting unknown]; [Date of meeting unknown]. Clemson, SC: Appalachian Society of American Foresters. 99 p.

Burrough, E.R., Jr.; Froehlich, H.A. 1972. Effects of forest fertilization on water quality in two small Oregon watersheds. Tech. Note 159. Portland, OR: U.S. Department of the Interior, Bureau of Land Management. 8 p.

Chamberlain, T.W.; Harr, R.D.; Everest, F.H. 1991. Timber harvesting, silviculture, and watershed processes. Influences of forest and rangeland management on salmonid fishes and their habitats. Spec. Publ. 19. Bethesda, MD: American Fisheries Society: 181–205.

Clayton, J.L.; Kennedy, D.A. 1985. Nutrient losses from timber harvest in the Idaho batholith. Soil Science Society of America Journal. 49: 1041–1049.

Corbett, E.S.; Heilman, R.E. 1975. Effects of management practices on water quantity and quality: New Jersey municipal watersheds. GTR–NE–13. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 196 p.

Corbett, E.S.; Lynch, J.A.; Sopper, W.E. 1975. Forest-management practices as related to nutrient leaching and water quality. In: Proceedings of a conference on non-point sources of water pollution; 1975 May 1–2; Blacksburg, VA. Blacksburg, VA: Virginia Water Resouces Research Center: 157–173.

Corbett, E.S.; Spencer, W. 1975. Effects of management practices on water quantity and quality. GTR–NE–13. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 25–31.

Currier, J.B. 1980. Nutrients. In: An approach to water resources evaluation of non-point silvicultrual sources (a procedural handbook). USEPA–600/8/80–012. Washington, DC: U.S. Environmental Protection Agency. 900 p. Chapter 10.

Davis, M.B. 1976. Pleistocene biography of temperate deciduous forests. Geoscience Manuscript. 13: 113–126.

Douglass, J.E.; Swank, W.T. 1975. Effects of management practices on water quality and quantity: Coweeta Hydrologic Laboratory, NC. Gen. Tech. Rep. NE–13. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 1–13.

Dunne, T.; Leopold, L. 1979. Water in environmental planning. San Francisco: W.H. Freeman and Company. 818 p.

Falletti, D.A. 1977. Sediment production in wildland environments: a review. In: Soil erosion: prediction and control. Spec. Publ. 21. West Lafayette, IN: Soil Conservation Society of America. 393 p.

Federer, C.A.; Hornbeck, J.W.; Tritton, L.M. [and others]. 1989. Long-term depletion of calcium and other nutrients in Eastern U.S. forests. Environmental Management. 13: 593–601.

Feller, M.C.; Kimmins, J.P. 1984. Effects of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. Water Resources Research. 20: 29–40.

Fenn, M.E.; Poth, M.A.; Aber, J.D. [and others]. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecological Applications. 8: 706–733.

Fredriksen, R.L. 1965. Sedimentation after logging road construction in a small western Oregon watershed. In: Proceedings of the Federal interagency sediment conference; 1963 January 28–February 1; Jackson, MS. Misc. Publ. 970. Washington, DC: U.S. Government Printing Office: 56–59.

Fredriksen, R.L. 1971. Comparative chemical quality—natural and disturbed streams following logging and slash burning. In: Krygier, J.T.; Hall, J.D., eds. Symposium on forest land use and the stream environment; 1970 October 19–21; Corvallis, OR. Corvallis, OR: Oregon State University Press. 252 p.

Fredriksen, R.L.; Moore, D.G.; Norris, L.A. 1975. Impact of timber harvest, fertilization, and herbicide treatment on stream water quality in western Oregon and Washington. In: Bernier, B.; Winget, C.H., eds. Forest soils and forest land management: Proceedings of the 4th North American forest soils conference; 1973 August; Quebec, Canada. Quebec: Laval University Press. 675 p.

Harr, R.D.; Fredriksen, R.L. 1988. Water quality after logging small watersheds within the Bull Run watershed, Oregon. Water Resources Bulletin. 24(5): 1103–1111.

Harr, R.D.; Fredriksen, R.L.; Rothacher, J. 1979. Changes in streamflow following timber harvest in southwestern Oregon. Res. Pap. PNW–249. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest Experiment Station. 22 p.

Heede, B.H.; King, R.M. 1990. State-of-the-art timber harvest in an Arizona mixed conifer forest has minimal effect on overland flow and erosion. Hydrological Sciences. 35: 623–635.

Hewlett, J.D. 1979. Forest water quality: an experiment in harvesting and regenerating Piedmont forest land. Athens, GA: University of Georgia Printing Office. 28 p. [As cited by Swank, W.T.; DeBano, L.F.; Nelson, D. 1989. Effects of timber management practices on soil and water. The scientific basis for silvicultural management decisions in the National Forest System. In: Burns, R.M., ed. Gen. Tech. Rep. WO-55. Washington, DC: U.S. Department of Agriculture, Forest Service: 79–106].

Hewlett, J.D.; Fortson, J.C. 1982. Stream temperature under an inadequate buffer strip in the Southeastern Piedmont. Water Resources Bulletin. 18(6): 983–988.

Holtby, L.; Newcombe, C.P. 1982. A preliminary analysis of logging related temperature changes in Carnation Creek, B.C. In: Hartman, G.F., ed. Proceedings of the Carnation Creek workshop; [Date of meeting unknown]; Victoria, BC, Canada: Ministry of Forests and Lands: 81–99.

Hornbeck, J.W.; Federer, C.A. 1975. Effects of management practices on water quality and quantity: Hubbard Brook Experimental Forest, New Hampshire. Gen. Tech. Rep. NE–13. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 68–75.

Hornbeck, J.W.; Martin, C.W.; Pierce, R.S. [and others]. 1987. The northern hardwood forest ecosystem: ten years of recovery from clearcutting. Res. Pap. NE–RP–596. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 21 p.

Hostetler, S.W. 1991. Analysis and modeling of long term stream temperatures on the Steamboat Creek Basin, Oregon: implications for land use and fish habitat. Water Resources Bulletin. 27(4): 637–647.

Ice, G.G. [In press (a)]. Stream temperature and dissolved oxygen: the Alsea watershed study. In: Stednick, J.D.; ed. The Alsea watershed: hydrological and biological responses to temperate coniferous forest practices. New York: Springer-Verlag.

Ice, G.G. [In press (b)]. Streamflow and water quality: what does the science show about clearcutting in western Oregon. In: Proceedings of a symposium: clearcutting in western Oregon: what does science show? 1999 November 3; Corvallis, OR. Corvallis, OR: Oregon State University.

Johnson, D.W.; Kelly, J.M.; Swank, W.T. [and others]. 1988. The effects of leaching and whole tree harvest on cation budgets of several forests. Journal of Environmental Quality. 17(3): 418–424.

Johnston, R.S. 1984. Effect of small aspen clearcuts on water yield and water quality. Res. Pap. INT–333. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 9 p.

Klock, G.O. 1971. Streamflow nitrogen loss following forest erosion control fertilization. Res. Note PNW–169. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 9 p.

Kochenderfer, J.N.; Aubertin, G.M. 1975. Effects of mangement practices on water quality and quantity: Fernow Experimental Forest, West Virginia. GTR–NE–13. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 14–24.

Krammes, J.S.; Burns, D.M. 1973. Road construction on Caspar Creek watersheds—10-year report on impact. Res. Pap. PSW–03. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 10 p.

Leaf, Charles F. 1974. A model for predicting erosion and sediment yield from secondary forest road construction. Res. Note RM–274. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 4 p.

Levno, A.; Rothacher, J. 1969. Increases in maximum stream temperature of the slash burning in a small experimental watershed. Res. Note PNW–16. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest Experiment Station. 13 p.

Likens, G.E.; Bormann, F.H.; Johnson, N.M. [and others]. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed ecosystem. Ecological Monograph. 40(1): 23–47.

Limerinos, J.T. 1978. Evaluation of thermograph data for California streams. U.S. Geol. Surv. Water Resour. Invest. 78–66. Menlo Park, CA: U.S. Geological Survey. 28 p.

Lynch, J.A.; Sopper, W.E.; Corbett, E.S.; Amard, D.W. 1975. Effects of management practices on water quantity and quality: the Penn State experimental watersheds. Gen. Tech. Rep. NE–13. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 32–46.

MacDonald, L.; Smart, A.W.; Wissmar, R.C. 1991. Monitoring guidelines to evaluate the effects of forestry practices on streams in the Pacific Northwest and Alaska. USEPA/910/9–91–001. Seattle: U.S. Environmental Protection Agency. 166 p.

Malueg, K.W.; Powers, C.F.; Krawczyk, D.F. 1972. Effects of aerial forest fertilization with urea pellets on nitrogen levels in a mountain stream. Northwest Science. 46(1): 52–58.

Mann, L.K.; Johnson, D.W.; West, D.C. [and others]. 1988. Effects of whole -tree and stem only clearcutting on post harvest hydrologic losses, nutrient capital and regrowth. Forest Science. 34: 412–428.

Martin, C.W.; Harr, R.D. 1989. Logging of mature Douglas-fir in western Oregon has little effect on nutrient output budgets. Canadian Journal of Forest Research. 19: 35–43.

Martin, C.W.; Noel, D.S.; Federer, C.A. 1984. Effects of forest clearcutting in New England on stream chemistry. Journal of Environmental Quality. 13: 204–210.

McClurkin, D.C.; Duffy, P.D.; Ursic, S.J.; Nelson, N.S. 1985. Water quality effects of clearcutting Upper Coastal Plain loblolly pine plantations. Journal of Environmental Quality. 14: 329–332.

McClurkin, D.C.; Duffy, P.D.; Ursic, S.J.; Nelson, N.S. 1987. Changes in forest floor and water quality following thinning and clearcutting of 20-year-old pine. Journal of Environmental Quality. 16: 237–241.

Meehan, W.R.; Lotspeich, F.B.; Muelle, E.W. 1975. Effects of forest fertilization in two southeast Alaska streams. Journal of Environmental Quality. 4(1): 50–53.

Megahan, Walter F. 1975. Sedimentation in relation to logging activities in the mountains of central Idaho. In: Present and prospective technology for predicting sediment yields and sources. Agric. Res. Serv. Rep. ARS–40. Beltsville, MD: U.S. Department of Agriculture, Agricultural Research Service: 74–82.9

Miller, E.L.; Beasley, R.S.; Lawson, E.R. 1988. Forest harvest and site preparation effects on erosion and sedimentation in the Ouachita Mountains. Journal of Environmental Quality. 17: 219–225.

Miller, J.H.; Newton, M. 1983. Nutrient losses from disturbed forest watersheds in Oregon's coast range. Agro-Ecosystems. 8: 153–167.

Moore, D.G. 1971. Fertilization and water quality. In: Proceedings of the annual meeting of the western reforestation coordination committee; [Date of meeting unknown]; [Location of meeting unknown]. Portland, OR: Western Forestry and Conservation Associates: 28–35.

Moore, D.G. 1975. Effects of forest fertilization with urea on water quality —Quilcene Ranger District, Washington. Res. Note PNW–241. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 9 p.

NCASI. 1999a. Scale considerations and the detectability of sedimentary cumulative watershed effects. NCASI Tech. Bull. 776. New York: National Council for Air and Stream Improvement. 227 p.

NCASI. 1999b. Water quality effects of forest fertilization. NCASI Tech. Bull. 782. New York: National Council for Air and Stream Improvement. 53 p.

Patric, J.H.; Evans, J.O.; Helvey, J.D. 1984. Summary of sediment yield data from forested land in the United States. Journal of Forestry. 82: 101–104.

Pierce, R.S.; Martin, C.W.; Reeves, C.C. [and others], eds. 1972. Nutrient losses from clearcuttings in New Hampshire. In: National symposium on watersheds in transition. 1972 June 19–22; Fort Collins, CO. Am. Water Resour. Assoc. Proc. Ser. 14. Urbana, IL: American Water Resources Association: 285–295.

Pye, J.; Vitousek, P. 1985. Soil and nutrient removals by erosion and windrowing at a Southeastern U.S. Piedmont site. Forest Ecology and Management. 11: 145–155.

Quigley, T.M. 1981. Estimating contribution of overstory vegetation to stream surface shade. Wildlife Society Bulletin. 9(1): 22–27.

Renfro, G.W. 1975. Use of erosion equations and sediment delivery ratios for predicting sediment yield. In: Present and prospective technology for predicting sediment yields and sources. Agric. Res. Serv. Rep. ARS–40. Beltsville, MD: U.S. Department of Agriculture, Agricultural Research Service: 33–45.

Rense, J.D.; Stottlemeyer, R.; Troendle, C.A. 1997. Effect of clearcutting on nutrient fluxes in a subalpine forest at Fraser, Colorado. Hydrology and Earth System Sciences. 1(2): 333–344.

Riggan, P.J.; Lockwood, R.N.; Lopez, E.N. 1985. Deposition and processing of airborne nitrogen pollutants in Mediterranean-type ecosystems of southern California. Environmental Science and Technology. 19: 781–789.

Rishel, G.B.; Lynch, J.A.; Corbett, E.S. 1982. Seasonal stream temperature changes following forest harvest. Journal of Environmental Quality. 11: 112–116.

Robichand, P.R.; Waldrop, T.A. 1994. A comparison of surface runoff and sediment yields from low- and high-severity site preparation burns. Water Resoures Bulletin. 30(1): 27–34.

Rothwell, R.L. 1971. Watershed management guidelines for logging and road construction. Ottawa, Ontario, Canada: Canadian Forest Service, Forest Res. Lab. [22 p.].

Silsbee, D.; Larson, G. 1982. Water quality of streams in the Great Smoky Mountain National Park. Hydrobiologia. 89: 97–115.

Snyder, G.G.; Haupt, H.F.; Belt, G.H., Jr. 1975. Clearcutting and buring slash alter quality of stream water in northern Idaho. Res. Pap. INT–168. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 34 p.

Sollins, P.; Grier, C.C.; McCorison, F.M. [and others]. 1980. The internal element cycles of an old growth Douglas fir ecosystem in western Oregon. Ecological Monographs. 50: 261–285.

Sopper, W.E. 1975. Effects of timber harvesting and related management practices on water quality in forested watersheds. Journal of Environmental Quality. 4(1): 24–29.

State of Idaho, Department of Health and Welfare. 1987. State of Idaho forest practices water quality management plan. Water Qual. Rep. Boise, ID: Idaho Department of Health and Welfare. 134 p.

Stednick, J.D. 1987. Potential for subalpine forest management practices on sediment production. In: Management of subalpine forests: building on 50 years of research. Gen. Tech. Rep. 149. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest Experiment Station: 95–100.

Stednick, J.D. 1991. Wildland water quality sampling and analysis. San Diego, CA: Academic Press. 217 p.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology. 176: 79–95.

Stone, E. 1973. The impact of timber harvest on soils and water. Report of the President's Advisory Panel on timber and the environment. Washington, DC: U.S. Government Printing Office: 427–467.

Stottlemeyer, R. 1987. Natural and anthropic factors as determinants of long-term stream water chemistry. In: Troendle, C.A.; Kaufmann, M.R.; Hamre, R.H.; Winokur, R.P., eds. Management of subalpine forests: building on 50 years of research. Gen. Tech. Rep. RM–149. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest Experiment Station: 86–94.

Stuart, G.; Dunshie, D. 1976. Effects of timber harvest on water chemistry. Hydrol. Pap. [Milwaukee, WI]: U.S. Department of Agriculture, Forest Service, Eastern Region. 34 p.

Swank, W.T. 1988. Stream chemistry responses to disturbance. In: Swank, W.T.; Crossley, D.A., eds. Forest hydrology and ecology at Coweeta. New York: Springer-Verlag: 339–358.

Swank, W.T.; DeBano, L.F.; Nelson, D. 1989. Effects of timber management practices on soil and water. In: Burns, R.L., tech. comp. The scientific basis for silvicultural and management decisions in the National Forest System. Gen. Tech. Rep. WO–55. Washington, DC: U.S. Department of Agriculture, Forest Service. 180 p.

Swank, W.T.; Johnson, C.E. 1994. Small catchment research in the evaluation and development of forest management practices. In: Biogeochemistry of small catchments: a tool for environmental research. SCOPE. New York: John Wiley. 419 p. Chapter 17.

Swift, F.W., Jr.; Messer, J.B. 1971. Forest cuttings raise temperatures of small streams in the Southern Appalachians. Journal of Soil and Water Conservation. 26: 11–116.

Swift, L., Jr. 1988. Forest access roads: design, maintenance, and soil loss. In: Swank, W.T.; Crossley, D.A., eds. Forest hydrology and ecology at Coweeta. New York: Springer-Verlag: 325–338.

Tiedemann, A.R. 1973. Stream chemistry following a forest fire and urea fertilization in north central Washington. Res. Note PNW–203. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 20 p.

Tiedemann, A.R.; Klock, G.O. 1973. First-year vegetation after fire, reseeding, and fertilization on the Entiat Experimental Forest. Note PNW–195. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 6 p.

Tiedemann, A.R.; Quigley, T.M.; Anderson, T.D. 1988. Effects of timber harvest on stream chemistry and dissolved nutrient losses in northeast Oregon. Forest Science. 2: 344–358.

Ursic, S.J. 1970. Hydrologic effects of prescribed burning and deadening upland hardwoods in northern Mississippi. Res. Pap. SO–54. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 15 p.

U.S. Department of Agriculture, Forest Service. 1980. An approach to water resources evaluation of nonpoint silvicultural sources (WRENS). EPA–600/8–80–012. Springfield, VA: U.S. Environmental Protection Agency. 900 p.

Van Lear, D.H.; Douglass, J.E.; Cox, S.K.; Augspurger, M.K. 1985. Sediment and nutrient export in runoff from burned and harvested pine watersheds in the South Carolina Piedmont. Journal of Environmental Quality. 14(2): 169–174.

Verry, E.S. 1972. Effect of aspen clearcutting on water yield and quality in northern Minnesota. In: Csallany, S.D.; McLaughlin, T.G.; Striffler, W.D., eds. National symposium on watersheds in transition; 1972 June 19–22; Fort Collins, CO. Urbana, IL: American Water Resources Association: 276–284.

Ward, T.J.; Baker, M.B. 1984. Sediment from managed pine watershed in northern Arizona. In: Proceedings of the specialty conference of the irrigation and drainage division of the American society of civil engineers; [Date of meeting unknown]; Flagstaff, AZ. Flagstaff, AZ: American Society of Civil Engineers: 552–557.

Chapter 11

Forest Succession

Wayne Swank[1]

Introduction

The effects of forest management activities on water quality are generally of the greatest magnitude in the first several years after disturbance. However, during long-term succession and regrowth of forest ecosystems, changes in physical, chemical, and biological parameters of streams may occur.

Nutrients

Issues and Risks

After a forest disturbance such as harvesting or fire, nutrient levels in streams may be elevated during early successional stages until the forest matures (see chapter 10). Nitrate concentrations can be elevated for a few to many years depending upon whether the watershed is nitrogen limited or saturated (see chapter 3 for discussion of nitrogen saturation).

Findings from Studies

Changes in stream inorganic chemistry and sediment yield were observed over a 20-year period after clearcutting by cable logging of a 146-acre [58-hectare (ha)] Southern Appalachian watershed (Swank and others, in press). Stream nutrient concentrations and fluxes showed small increases after harvest, and responses were largest the third year after treatment. Nitrate-nitrogen (NO_3^{-1}) was an exception. The initial increase in nitrate was from <0.1 milligrams (mg) per liter to 0.8 mg per liter (fig. 11.1) and increased net nitrogen export of 1.16 pounds per acre [1.3 kilograms (kg) per hectare] the third year after harvest. However, later in succession (15 to 20 years), nitrate concentrations exceeded values observed the first several years after clearcutting. This response is partially attributed to reduction in nitrate uptake due to vegetation mortality, changes in species composition, and nitrogen release from decomposition of woody plants.

Other long-term research in eastern forested watersheds (Edwards and Helvey 1991, Swank and Vose 1997) shows that as forests mature, less nitrogen is retained in the watershed and stream nitrate concentrations increase. These long-term studies support findings of shorter term stream chemistry surveys. A survey of streamwater chemistry in 57 watersheds along successional and elevational gradients was conducted in the White Mountains of New Hampshire (Vitousek 1977). Differences in successional status among watersheds were found to be important in controlling nitrate and potassium concentration. Streams draining old-aged forests had higher concentrations of nitrate, potassium, and other solutes than did streams draining intermediate-aged forests at the same elevation. Spruce-fir (*Picea* spp.-*Abies* spp.) watersheds with no record of logging had streamwater nitrate concentrations of about 3 mg per liter, while spruce-fir watersheds logged 30 years previously had nitrate concentrations <0.5 mg per liter.

Another survey of 38 streams draining partially or entirely clearcut watersheds was conducted in New England—(Martin and others 1985) on northern hardwood sites in New Hampshire, Maine, and Vermont; in central hardwood forests in Connecticut; and in coniferous forests in Maine and Vermont. Streams draining watersheds that had been partially or entirely clearcut in the previous 2 years were selected. There were no apparent changes in stream nutrient concentrations from many of the ecosystems, and the largest concentration increases were for nitrate, calcium, and potassium in northern hardwoods of New Hampshire. Inorganic nitrogen (nitrate plus ammonium) increased to an average of 2 mg per liter (Martin and others 1985). However, elevated solute concentrations appear to be short-lived, even in streams draining successional northern hardwood forests in New Hampshire (Hornbeck and others 1987). Moreover, early stream chemistry changes after clearcutting were considered insufficient to cause concern for public water supplies or for downstream nutrient loading (Martin and others 1985).

In the Pacific Northwest, forest-successional stage is not always a good predictor of nitrate concentration in streamwater. For example, at the H.J. Andrews Experimental Forest in Oregon, forest harvest increased annual nitrate

[1] Retired Project Leader, USDA Forest Service, Southern Research Station, Otto, NC.

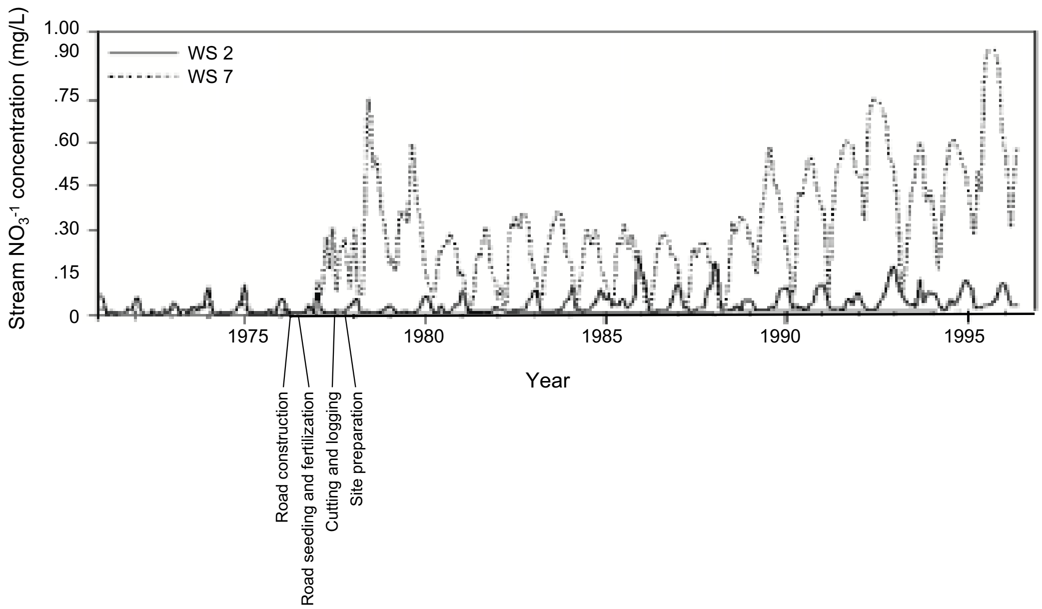


Figure 11.1—Mean monthly concentrations (flow weighted) of nitrate (NO_3^{-1}) in streamwater of a clearcut, cable-logged, hardwood-covered watershed (WS7) and an adjacent watershed (WS2) during calibration, treatment activities, and postharvest period, Coweeta Hydrologic Laboratory, North Carolina.

concentration from predisturbance levels of 0.001 mg per liter to 0.036 mg per liter (Martin and Harr 1989), but nitrate concentration returned to predisturbance levels within 6 years. Further, a 20-year postdisturbance record from a pair of treated and untreated watersheds at the experimental forest suggests that nitrate concentrations in streamwater remain very low in both watersheds once the clearcut watershed recovers from the immediate effects of disturbance.[2] At the H.J. Andrews Experimental Forest, the ecosystem is highly nitrogen-limited, and vegetation imprint on nitrogen fluxes may be overridden by rapid immobilization of any available nitrogen by soil microbiota.

An extensive synoptic water-quality assessment was conducted on numerous streams in the Great Smoky Mountain National Park in the Southern Appalachian Mountains (Flum and Nodvin 1995, Silsbee and Larson 1982). Concentrations of nitrate in streams draining watersheds that had been logged prior to park establishment were significantly lower (one-half) than the nitrate concentrations in unlogged watersheds at similar elevations.

The magnitude of stream nitrate concentrations associated with long-term forest succession depends on a number of factors, such as levels of atmospheric nitrogen deposition, the type and rapidity of forest regrowth, soil microbial activity, and soil physiochemical reactions. Stream nitrate levels rarely exceed 5 mg per liter and are below current drinking water standards. The nitrate, however, may contribute to stream acidification, particularly during spring snowmelt when nitrate concentrations peak in the Northeastern United States (Murdock and Stoddard 1992).

Reliability and Limitations of Findings

Existing evidence for changes in stream chemistry with forest succession is based upon well-established programs of long-term research and is quite reliable. However, findings are limited in scope to select forest ecosystems in the United States.

Limited evidence indicates that stream nitrate concentrations for older hardwood forests of the southern and central Appalachian regions are higher than for younger successional forests. However, site-specific research shows that nitrate levels can vary substantially even during early succession (first 20 years), although the general applicability of findings is unknown. Assessments of nitrate levels in streams draining successional forests in New England show mixed responses and appear to be ecosystem specific. Very limited information on stream nitrate is available for successional forests in the Pacific Northwest. Current findings

[2] Personal communication. 1999. Kristin Vanderbilt, Graduate Student. Oregon State University, Corvallis, OR.

indicate that elevated nitrate concentrations following clear-cutting are short-lived and return to predisturbance levels early in succession.

Research Needs

1. Long-term assessments of stream chemistry changes associated with forest succession are lacking for most major forest ecosystems in the United States. From a public drinking water perspective, synoptic stream nutrient surveys across a range of forest types and stand ages with known disturbance histories would greatly enhance planning information for managers.
2. There is a large knowledge gap in nutrient concentration changes associated with storm runoff events. Such information is most important where water supplies are derived from forested headwaters with rapid streamflow responses to precipitation, e.g., watersheds with shallow soils, steep slopes, intense rainfall, and rapid snowmelt.

Sediment

Issues and Risks

Stream sediment may also exhibit long-term dynamics after forest disturbance. Logging roads associated with harvesting activities are frequently the major source of sediment to streams and are a potential legacy to consider when evaluating sources of sediment in drinking water (see chapters 3, 9).

Findings from Studies

A synthesis of long-term sediment yield responses following forest watershed disturbances is provided by Bunte and MacDonald (1999). Based on studies in Oregon and New Hampshire, they identify three kinds of potential responses in postdisturbance sediment yields:

1. Sediment yields remain high for a number of years after disturbance due to a large sediment pulse to the stream from a storm or other disturbance. That is, sediment from upstream storage areas or destabilized hillslopes and channels continues to be released;
2. Sediment yields decline below average annual yields after disturbance when sediment storage is depleted by a major sediment transport event; and
3. Sediment yields rapidly return to predisturbance conditions because excess material has moved through the system.

Recent findings in the Southern Appalachians provide an example of the first type of response where sediment yield remains high for a number of years during forest succession (Swank and others, in press). A cable-logged, clearcut watershed required only three contour access roads because logs could be yarded 1,000 feet (305 meters) with the cable system. Record storms (15 inches or 38 centimeters) in the last 2 weeks of May 1976, prior to grass establishment, eroded both unstable soil and hydroseeded materials from the roads. Roads were the source of elevated sediment yield as illustrated by soil loss measured at a gaging station in the stream immediately below a road crossing in the middle of the catchment (fig.11.2A). In those 2 weeks of May, sediment yield was nearly 55 tons [50 metric tonnes (Mg)] from 0.21 acre (0.085 ha) of road contributing area (roadbed, cut, and fill). In the ensuing period of road stabilization and minimum use (June to December 1976), soil loss was low but accelerated again briefly during the peak of logging activities (fig. 11.2A). In the next year, soil loss below the road declined to baseline levels.

The pattern of sediment yield at the base of the second-order stream (fig. 11.2B, gaging site) draining the watershed was different from the pattern of sediment loss from the roads. Following an initial pulse of sediment export from the watershed, sediment yield remained substantially elevated during and after logging. In the 3-year period between 1977–80, the cumulative increase in sediment yield was 240 tons (218 Mg) (fig. 11.2B). During the next 10 years, sediment yield declined with a cumulative increase in export of 240 tons (218 Mg). The rate of sediment yield over the 5- to 15-year period after disturbance was about 300 lb per acre per year (336 kg per hectare per year), or 50 percent above pretreatment levels. The long-term sediment yield data illustrate a lag or delay between pulsed sediment inputs to a stream and the routing of sediments through the stream channels. In the absence of significant additional sources of sediment to streams on the watershed, annual sediment yield at the base of the watershed was still substantially above predisturbance levels at least 15 years later. Thus, there appears to be a continual release of sediment from upstream storage that was primarily deposited from road crossings of streams during exceptionally severe storms.

Reliability and Limitations of Findings

Few studies have documented the long-term effects of management practices on sediment yield. As pointed out in chapter 10, increases in sediment yields from timber management activities are typically considered to be short-lived. However, unique conditions during management can lead to elevated stream sediment later in forest succession. The importance of this process is site-specific and requires

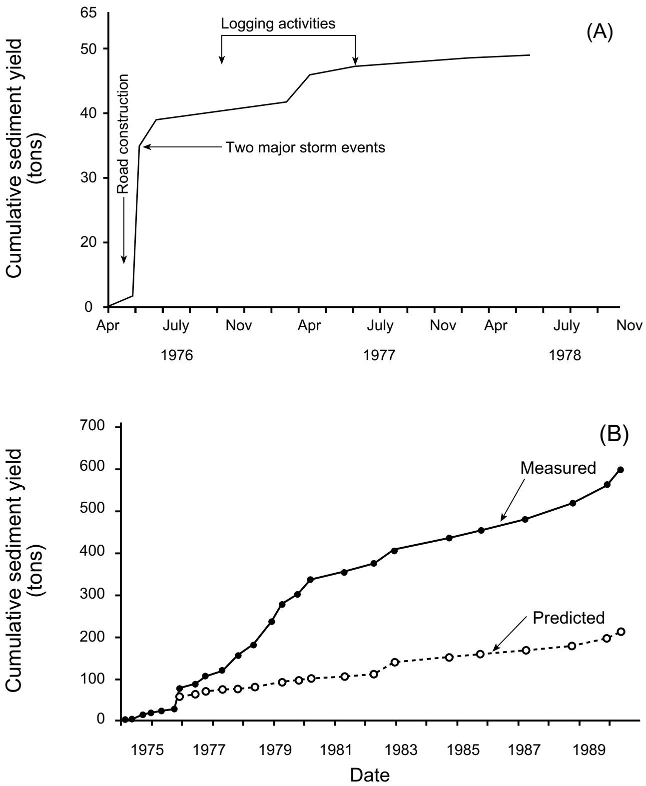


Figure 11.2—Cumulative sediment yield measured on a clearcut, cable-logged, hardwood-covered watershed: (A) in one of the first-order streams below a logging road during the first 32 months after treatment and (B) in the ponding basin of the second-order stream at the gaging site during 15 years after treatment. Predicted values are based on pretreatment calibration of sediment yield with an adjacent control watershed, Coweeta Hydrologic Laboratory, North Carolina.

that each stream be evaluated to assess the legacy of past management practices on current levels of stream sedimentation.

Research Need

Recommendations for future research related to this topic are given in chapter 10.

Key Points

In the long term, forest harvesting practices alone may have little deleterious impact on stream sediment and chemistry, which are of primary concern in drinking water. However, other past and present land uses affect present sediment and nitrate concentrations in streams. Sediment and nitrate yields associated with early successional development of forest may be in addition to yields from other past and present land uses. It is important to consider successional impacts along with the cumulative impacts of other past and present land uses across the landscape when assessing impacts of land management on drinking water sources.

Literature Cited

Bunte, K.; MacDonald, L.H. 1999. Scale considerations and the detectability of sedimentary cumulative watershed effects. Tech. Bull. 776. Research Triangle Park, NC: National Council for Air and Stream Improvement. 327 p.

Edwards, P.J.; Helvey, J.D. 1991. Long-term ionic increases from a control Appalachian forested watershed. Journal of Environmental Quality. 20: 250–255.

Flum, T.; Nodvin, S.C. 1995. Factors affecting streamwater chemistry in the Great Smoky Mountains, U.S.A. Water, Air and Soil Pollution. 85: 1707–1712.

Hornbeck, J.W.; Martin, C.W.; Pierce, R.S. [and others]. 1987. The northern hardwood forest ecosystem: ten years of recovery from clearcutting. Res. Pap. NE–596. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 30 p.

Martin, C.W.; Harr, R.D. 1989. Logging of mature Douglas-fir in western Oregon has little effect on nutrient output budgets. Canadian Journal of Forest Research. 19: 35–43.

Martin, C.W.; Noel, D.S.; Federer, C.A. 1985. Clearcutting and the biogeochemistry of streamwater in New England. Journal of Forestry. 83(11): 686–689.

Murdock, P.S.; Stoddard, J.L. 1992. The role of nitrate in the acidification of streams in the Catskill Mountains of New York. Water Resources Research. 28(10): 2707–2720.

Silsbee, D.G.; Larson, G.L. 1982. Water quality of streams in the Great Smoky Mountains National Park. Hydrobiologia. 89: 97–115.

Swank, W.T.; Vose, J.M. 1997. Long-term nitrogen dynamics of Coweeta forested watersheds in the Southeastern United States of America. Global Biogeochemical Cycles. 11(4): 657–671.

Swank, W.T.; Vose, J.M.; Elliott, K.J. [In press]. Long-term hydrologic and water quality responses following commercial clearcutting of mixed hardwoods on a Southern Appalachian catchment. Forest Ecology and Management.

Vitousek, P.M. 1977. The regulation of element concentrations in mountain streams in the Northeastern United States. Ecological Monographs. 47: 65–87.

Chapter 12

Fire Management

Johanna D. Landsberg and Arthur R. Tiedemann[1]

Introduction

The effect of wildfire on drinking water was graphically demonstrated when the Buffalo Creek fire in Colorado in 1996 was followed by heavy rains, forcing municipal water supplies to shut off, one of Denver's water treatment plants to close, months to be spent cleaning a water-supply reservoir, and the Coors Brewing Company to bring in water by truck (Illg and Illg 1997).

Fire, both wild and prescribed, has the potential to alter physical, chemical, and biological properties of surface water that originates from burned wildland areas. Nonpoint-source pollution from wildland after fire can impair the suitability of water for drinking and other purposes. New plans for widespread use of prescribed fire to solve forest health problems create an urgent need to fully understand the water-quality consequences of increasing the occurrence of fire. Fire management activities (like retardant application, fireline construction, and postfire rehabilitation) also have potential effects on water quality.

The most important effects of fire on drinking water source quality include sediment and turbidity or both, water temperature, and increased nutrients in streamflow. In this chapter, we review results of research on the response of the above water-quality variables to fire, fire management activities, and fire rehabilitation measures. Much of the information comes from reports on wildfires. We would expect the magnitude of streamwater-quality changes after prescribed fire to be less than those observed after wildfires and some broadcast slash burns. It is unlikely that prescribed fire would consume as much forest floor and understory, or kill as much overstory, as would a wildfire because prescribed fires are usually conducted under conditions deliberately chosen to produce burns of low severity.

[1] Research Fire Ecologist/Soil Scientist and Scientist Emeritus, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, Wenatchee, WA, respectively.

Sediment and Turbidity

Issues and Risks

Suspended sediment is the major nonpoint-source pollution problem in forests (Society of American Foresters 1995). Beschta (1990) reported that sediment and turbidity are the most significant water-quality responses associated with fire. Turbidity has no direct health effects but can interfere with disinfection and provides a medium for microbial growth. Thus, it may indicate the presence of microbes (U.S. EPA 1999). See chapter 2 for more discussion on the effects of sediment on drinking water.

Findings from Studies

To understand research findings about sediment production and its impacts, one must be familiar with the units of measurement in which sediment is reported. Suspended sediment is particles carried in suspension and is measured by filtering and drying a known volume of water. Suspended sediment is expressed in parts per million (ppm), or as turbidity in nephelometric turbidity units (NTU's), which is a measure of the cloudiness of the water. These methods measure different characteristics of water, and it is difficult to correlate the results of one method with results of the other. The standard turbidity method (U.S. EPA 1999) uses NTU's. We found only two studies of fire effects that reported results in NTU's [equivalent to Jackson turbidity units (JTU's)] from American Public Health Association (1976) (table 12.1); all others reported sediment in parts per million (table 12.2). Beschta (1980) found that a relationship between suspended sediment and turbidity can be established but that the relationship differs significantly among watersheds. He suggested that the relationship must be established on a watershed-by-watershed basis. Recognizing this difficulty, Helvey and others (1985) determined the relationship between sediment in parts per million and turbidity in NTU's for three catchments in northcentral Washington and found the relationship to be strong (Helvey and others 1985). With this strong relationship and the equations developed, sediment measurements, in parts per million, can be converted to turbidity measurements, in

NTU's (fig. 12.1). This relationship has not been tested in other geographic areas or plant community types, so caution is advised when applying it beyond its original limits. It does point out the need to use a standard method or to establish relations between suspended sediment and turbidity for each watershed or stream system in question.

Our interest here is on the effects of fire on sediment measured in NTU's. Wright and others (1976, 1982) found that slope plays an important role in the amount of turbidity in streamflow after broadcast burning oak-juniper (*Quercus* spp.-*Juniperus* spp.) watersheds in central Texas. Turbidity changes (table 12.1) after burning were most pronounced in the steepest watersheds, with levels reaching 230 JTU's.

Studies of suspended sediment (table 12.2) show that the range of the prefire or control values is 1 to 26 ppm. Values obtained after fires indicate that fire has a profound effect on sediment movement.

Table 12.1—Water turbidity, in Jackson turbidity units (equivalent to nephelometric turbidity units), after fire alone or in combination with other treatments

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - Jackson turbidity units - -		
Prescribed fire, pile, and burn	Juniper	Central Texas			Wright and others 1976
		3 to 4% slope	12	12	
		8 to 20% slope	20	53	
		37 to 61% slope	12	132	
Pile and burn	Juniper	Central Texas	12	162	Wright and others 1982
Pile, burn, and seed			12	72	

Table 12.2—Suspended sediment concentration in streamflow after fire alone or in combination with other treatments

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - Parts per million - - - - -		
Wildfire	Taiga	Interior Alaska	3.7 –10.6	2.6 – 6.0	Lotspeich and others 1970
Clearcut, slash broadcast burned	Douglas-fir	Western Oregon	2	56 –150	Fredriksen 1971
Wildfire	Ponderosa pine, Douglas-fir	Eastern Washington	Not known	1,200[a]	Helvey 1980
Pile, burn	Juniper	Central Texas	1.1	3.7	Wright and others 1982
Pile, burn, and seed			1.0	3.7	
Prescribed fire	Loblolly pine plantation	Upper Piedmont, South Carolina	26	33	Douglass and Van Lear 1983
Wildfire	Lodgepole pine, Douglas-fir, ponderosa pine, western larch	Glacier National Park, MT	< 3	15 – 32	Hauer and Spencer 1998

[a] Maximum value attained.

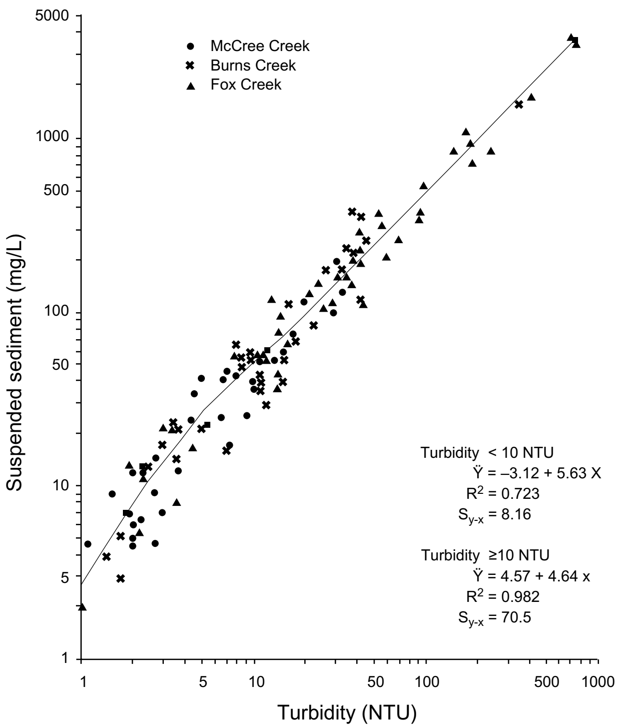


Figure 12.1—Relationship between turbidity in nephelometric turbidity units (NTU) and suspended sediment parts per million (ppm) (Helvey and others 1985).

Sediment yield has been measured in pounds per acre per year in many studies because of the concern for soil loss after fire. Sediment yield varies widely as a consequence of fire or forest harvest and fire (table 12.3). This variability reflects numerous interacting factors: geology, soil, slope, vegetation, fire characteristics, treatment combinations, weather patterns, and climate.

In the research we reviewed, sediment yield from pretreatment or control areas ranged from as low as 3 pounds per acre per year [3.36 kilograms (kg) per hectare per year] to as high as 12,500 pounds per acre per year [14 metric tonnes (Mg) per hectare per year] (table 12.3). Postburn sediment yield ranged from as low as 12 pounds per acre per year (13.5 kg per hectare per year) to as high as 98,160 pounds per acre per year (110 Mg per hectare per year). The lower values generally were associated with flatter land and lower severity fires. The higher values resulted from more severe fires on steeper slopes and from fires on areas with soils formed from decomposing granite, which erode readily.

When fire is used to convert brush to grass, it can have an unintended side effect: mass wasting, which can affect water quality. Work in California established the susceptibility of steep slopes to mass soil movement following conversion of brush to grass (Bailey and Rice 1969). These mass soil movements produce long-lasting changes. In one study, these same effects occurred on steep, forested slopes; especially after severe fires (Robichaud and Waldrop 1994) (table 12.3). These sediment yields are sufficient to generate concern about water turbidity, which was not measured directly.

Burned areas are sometimes seeded to rapidly establish plants or are given other treatments to quickly stabilize the soil. Following severe wildfire, the Forest Service and other land managers sometimes implement Burn Area Emergency Rehabilitation (BAER) treatments to reduce the risk of high runoff and sediment flows to vulnerable installations downstream such as drinking water intakes and reservoirs. In a review of literature and monitoring reports, Robichaud and others (in press) found that the effectiveness of the most widely used BAER practice, contour-felled log barriers, had not been systematically studied. The second most used BAER practice, postfire broadcast seeding with grasses, has been studied and the majority of studies found that this treatment did not significantly reduce erosion during the critical first 2 years after fire (Robichaud and others 2000). Effectiveness of contour felling has not been tested, and reseeding with grasses is not a reliable technique for erosion control after severe wildfire. Additionally, when an area is seeded with nonnative grass species, native plant species may be effectively excluded leading to questions about long-term stability (Tiedemann and Klock 1976).

Firelines, particularly those that are created by bulldozers, are important potential sources of suspended sediment and turbidity in streams for several reasons. First, some firelines are constructed in urgent circumstances, without adequate time to consider stream protection. Thus, they may provide direct channels for sediment into streams. Second, firelines may be difficult to stabilize with vegetation because much of the nutrient-rich surface soil is cast aside. Hence, they are likely to be slow to revegetate with perennial vegetation. Information on revegetating and stabilizing firelines is very limited. Two studies found application of seed and fertilizer is an effective way to protect firelines (Klock and others 1975, Tiedemann and Driver 1983). Klock and others (1975) demonstrated that seeding firelines with several species of introduced and native grasses produced up to 85 percent foliar cover within 2 years. In their area of nitrogen- and sulfur-limited soils, starter fertilizer containing nitrogen and sulfur substantially improved plant foliar cover and was considered to be essential for successful seeding.

Table 12.3—Sediment yield after fire alone or in combination with other treatments

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			Pounds per acre per year		
Prescribed burn	Bluestem grasses	Upper Coastal Plain, northern Mississippi	74 – 84	1,542 – 5,759	Ursic 1969
Prescribed underburn	Post oak, hickories, blackjack oak	Upper Coastal Plain, northern Mississippi	148 – 300	868 – 1,179	Ursic 1970
Prescribed fire, pile, and burn	Juniper	Central Texas			Wright and others 1976
		3 to 4% slope	18	18	
		8 to 20% slope	146	504	
		37 to 61% slope	6	5,554	
Wildfire	Ponderosa pine	Northwestern Arizona	3	1,254	Campbell and others 1977
	Ponderosa pine, Douglas-fir	Eastern Washington	12 – 35	146 – 2,100	Helvey 1980, Helvey and others 1985
Prescribed underburn	Loblolly pine plantation	Upper Piedmont, South Carolina	19	20	Douglass and Van Lear 1983
Wildfire	Chaparral	Southern California	12,500	98,160	Wells 1986
Clearcut: broadcast burned, planted to Douglas-fir, wildfire	After clearcut: vegetation: tanoak, madrone, chinkapin, black oak, poison oak	Southern Oregon	80 [a]	55	Amaranthus 1989
Adjacent forest: wildfire	Douglas-fir overstory; tanoak, madrone, black oak understory		79	40	Amaranthus 1989
Clearcut and prescribed fire:	Oak spp., shortleaf pine	Northwestern South Carolina	Not known		Robichaud and Waldrop 1994
Low severity				12.1	
High severity				502	

[a] From October 13 to May 4, after September wildfire.

Temperature

Issues and Risks

Increases in streamwater temperature have important effects on aquatic habitat and stream and lake eutrophication. Eutrophication can adversely affect the color, taste, and smell of drinking water. See chapter 2 for temperature impacts on drinking water.

Findings from Studies

When riparian vegetation is removed by fire or other means, the stream surface is exposed to direct solar radiation, and stream temperatures increase (Levno and Rothacher 1969, Swift and Messer 1971). For example, clearcutting and slash burning increased stream temperatures by 13.0, 14.0, and 12.1 °F (7.2, 7.7, and 6.7 °C) in June, July, and August, with temperatures reaching a maximum of 75 °F (23.9 °C) in July (Levno and Rothacher 1969). Helvey (1972) found that during the first year after wildfire in eastern Washington, stream temperature increased 10 °F (5.6 °C). In southern Oregon, Amaranthus and others (1989) determined that temperatures increased 6, 11, and 18 °F (3.3, 6.1, and 10 °C), from a low temperature of 55 °F (12.8 °C) to a high temperature of 73 °F (22.8 °C) after a wildfire. These temperature changes have the potential to increase the rate of eutrophication if phosphate is present in abundance.

Chemical Water Quality

Several chemical constituents are likely to come from forest and rangeland burning. The primary ones of concern are nitrate (NO_3^{-1}) and nitrite (NO_2^{-1}). Sulfate, pH, total dissolved solids, chloride, iron, turbidity (discussed previously), and several other constituents can also be affected, as can color, taste, and smell (see chapter 2). Phosphate (P) can affect water quality because of its ability to affect color, taste, and smell by accelerating the eutrophication process.

To understand the influence of fire on water quality, it is important to understand some of the changes in plant, forest floor, and soil nutrients during and after the combustion process. Burning oxidizes organic material, resulting in direct loss of elements to the atmosphere as volatilized compounds above critical temperatures, as particulates are carried away in smoke, or elements are converted to oxides to the ash layer (DeBano and others 1998, Raison and others 1985, Tiedemann 1981) (fig. 12.2). Nitrogen, sulfur, and potassium are all susceptible to volatilization loss by burning (DeBano and others 1998, Raison and others 1985, Tiedemann 1987). Nitrogen is lost when temperatures reach

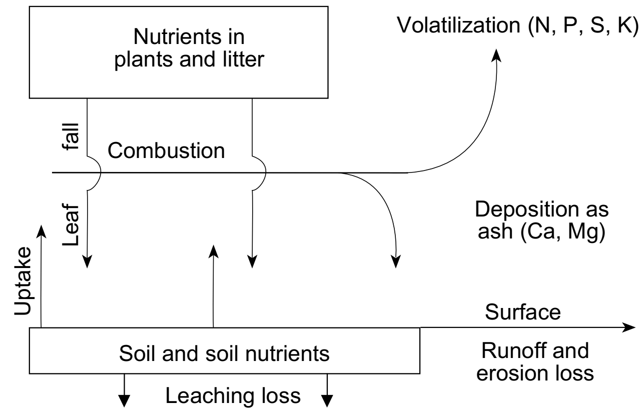


Figure 12.2—Possible pathways of plant- and litter-contained nutrients in response to combustion (Tiedemann 1981).

400 °F (204 °C) (DeBano and others 1998). At temperatures as low as 700 °F (371 °C), loss of sulfur can be substantial (Tiedemann 1987). As temperatures approach 1,475 °F (802 °C), virtually all nitrogen and sulfur are volatilized. At 1,430 °F (776 °C), phosphorus and potassium are volatilized. In ashes, relatively insoluble oxides of metallic cations, such as calcium, potassium, magnesium, and iron, react with water and carbon dioxide in the atmosphere and become more soluble (DeBano and others 1998, Tiedemann 1981) (fig. 12.2). This conversion increases potential for leaching loss of nutrients from the ash into and through the soil (DeBano and others 1998, Tiedemann 1981, Wells and others 1979). Nutrients in the ash are also susceptible to loss by surface erosion (Beschta 1990, DeBano and others 1998, Tiedemann 1981, Tiedemann and others 1979, Wells and others 1979).

The potential for increased nitrate in streamflow occurs mainly because of accelerated mineralization and nitrification in soils after burning (Covington and Sackett 1986, 1992; DeBano and others 1998; Vitousek and Melillo 1979), as well as reduced plant demand (Vitousek and Melillo 1979). This effect is short-lived, usually lasting only a year or so (Monleon and others 1997).

Transport of nutrients to streams occurs both during and after a wildland fire. Spencer and Hauer (1991) reported that the source of nitrogen in streamwater during a fire appears to be diffusion of smoke and gasses directly into the streamwater, and that the source of phosphorus in streamwater appears to be from the leaching of ash deposited directly into the stream. After a fire, nutrients from ash deposition move from the soil into streamwater when precipitation is adequate for percolation below the root zone, and when the capacity of vegetation for uptake or soil nutrient storage capacity, or both, are insufficient to retain mobile nutrients carried into the soil (Beschta 1990, DeBano and others 1998, Tiedemann and others 1979).

Issues and Risks

The issue is whether forest or rangeland fires degrade the quality of source water for public consumption by the introduction of additional chemical constituents. The risk is when these additional chemical constituents—from a fire or from fertilizer applied to establish vegetation in the burn area—are combined with chemical constituents already present, the source water supply may be degraded.

Findings from Studies

Immediately after a fire, the pH of streams may be affected by direct ash deposition. In the first year after fire, increased pH of the soil (Wells and others 1979) may also contribute to increased streamwater pH. In all the studies we evaluated (table 12.4), only one reported a notable increase in pH values. During the first 8 months after the Entiat fires in eastern Washington, Tiedemann (1973) detected transient pH values up to 9.5. Two days after fertilization, they detected a transient pH value of 9.2. In most studies pH values were little changed by fire and fire-associated events.

Nitrogen

The forms of nitrogen that are of concern in drinking water after fire are nitrate and nitrite. Values for nitrate generally increased after fire but not to a level of concern (table 12.5), except in nitrogen-saturated areas (see chapter 3). Stream nitrate responses to prescribed fire are generally lower than for wildfire. In an undisturbed ponderosa pine and Gambel oak or both (*P. ponderosa* Dougl. ex Laws. and *Q. gambelii* Nutt. or both) watershed in Arizona, Gottfried and DeBano (1990) found that a fire resulted in only slight, but significant, increases in nitrate (table 12.5). Measures to protect streams and riparian areas during prescribed burns with unburned buffers could minimize effects of fire on stream chemistry.

The most striking response of nitrate concentration in streamflow after wildfire (table 12.5) was observed in southern California (Riggan and others 1994). Moderate burning resulted in a maximum nitrate concentration of 9.5 ppm, while severe burning resulted in a maximum concentration of 15.3 ppm in streamflow, compared to 2.5 ppm in streamflow from an unburned control watershed. The concentration of 15.3 ppm is above maximum contaminant level for drinking water of 10 ppm (chapter 2, table 2.3). Chronic atmospheric deposition of nitrogen pollutants on these watersheds, which are east of Los Angeles, CA, have caused their soils to become nitrogen saturated. Beschta (1990) reached the same conclusion in his assessment of streamflow nitrate responses to fire and associated treatments. Fenn and others (1998) have discussed excess nitrogen in ecosystems in North America. These excess levels can lead to leaching of nitrate, which ultimately can find its way into streamwater (see chapter 3).

Fertilization after fire resulted in higher concentrations of nitrate than fire alone (table 12.5) (Tiedemann 1973, Tiedemann and others 1978). Nonetheless, Tiedemann (1973) concluded that neither fire nor nitrogen fertilization at levels less than 54 pounds per acre (60.5 kg per hectare) of elemental nitrogen would probably have adverse effects on nitrate concentrations in drinking water. Their research was done in an area with nitrogen-limiting soils. In areas experiencing nitrogen saturation, nitrogen fertilization may aggravate nitrate levels in water and is not likely to stimulate revegetation.

Nitrite was reported by itself, rather than in combination with nitrate, in only two studies that we found. At concentrations >1 ppm, nitrite can lead to serious illness in infants (chapter 2, table 2.3). At the Lexington Reservoir, Santa Clara County, CA, Taylor and others (1993) found nitrite levels of 0.03 ppm after the watershed above the reservoir was burned, while control levels were 0.01 ppm. Tiedemann (1973) reported that nitrite concentrations were below the levels of detection. The concentrations found do not appear to be a concern.

Fire retardants containing nitrogen have the potential to affect the quality of drinking water, but research on the application of retardants to streams has focused on the effects on fish and aquatic habitat (Buhl and Hamilton 1998; Gaikowski and others 1996; McDonald and others 1996, 1997; Norris and Webb 1989; Norris and others 1978). Several in vitro research projects evaluated the toxicity to stream organisms of some retardant formulations. The tested compounds were nonfoam retardants containing sulfate, phosphate, and ammonium compounds; a retardant containing ammonium and phosphate compounds; and two foam suppressant compounds (Buhl and Hamilton 1998; Gaikowski and others 1996; McDonald and others 1996, 1997). Concentrations of nitrate rose from 0.08 to 3.93 ppm after adding the nonfoam retardants. In addition, they found

Table 12.4—The pH in water after fire alone or in combination with other treatments usually remains fairly constant

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - - - - - *pH*	- - - - - - - - -	
Wildfire	Ponderosa pine, Douglas-fir	Eastern Washington	None given	7.2 – 8.5	Tiedemann 1973
Wildfire and N fertilization			None given	7.1 – 9.5[a]	
Wildfire and N fertilization	Mixed conifer, shrub	Central Sierra Nevada Mountains, California	~7.0 – 6.2[b]	~7.0 – 6.6[b]	Hoffman and Ferreira 1976
Pile, burn	Juniper	Central Texas			
		3 to 4% slope	7.3	7.3	Wright and others 1976
		8 to 20% slope	7.6	7.7	
		37 to 61% slope	7.4	7.7	
Wildfire	Pine, spruce, fir, aspen, birch[c]	Northeastern Minnesota lakes	6.2	6.1 – 6.3	Tarapchak and Wright 1977
Wildfire Wildfire and N fertilization	Ponderosa pine, Douglas-fir	Eastern Washington	7.4 – 7.6	7.4 – 7.6	Tiedemann and others 1978
Prescribed fire	Ponderosa pine	Central Arizona	6.2	6.4	Sims and others 1981
Pile, burn, and seed	Juniper	Central Texas	7.1	7.3	Wright and others 1982
Clearcut, slash broadcast burned	Western hemlock, western red cedar, Douglas-fir	Western British Columbia	6.8	7.8	Feller and Kimmins 1984
Yellowstone wildfires	Subalpine lake	Yellowstone Lake, Yellowstone National Park, WY	7.4	7.5	Lathrop 1994

~ = About or approximately.
[a] Transient pH value of 9.5 was observed second day after urea fertilization.
[b] From May to July during the summer following the August fire.
[c] Cited in Wright and Watts 1969.

Table 12.5—Maximum nitrate-nitrogen concentration in water after fire alone or in combination with other treatments

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - Parts per million - - - - -		
Clearcut, slash burned	Douglas-fir	Western Oregon	0.1	0.43	Fredriksen 1971
Wildfire	Ponderosa pine	Eastern Washington	.016[a]	.042	Tiedemann 1973
Wildfire and nitrogen fertilization	Douglas-fir		.005	.310[a]	
Wildfire	Mixed conifer, shrub	Central Sierra Nevada Mountains, California	~.6[a b]	~.12	Hoffman and Ferreira 1976
	Ponderosa pine	Northwestern Arizona	.086	.212	Campbell and others 1977
	Pine, spruce, fir, aspen, birch[c]	Northeastern Minnesota lakes	.17	.08 – .17	Tarapchak and Wright 1977
Wildfire	Ponderosa pine	Eastern Washington	< .016[a]	.56	Tiedemann and others 1978
Wildfire and nitrogen fertilization	Douglas-fir		< .016[a]	.54 –1.47	
Prescribed fire	Pine forest or not given	Lower Coastal Plain, South Carolina	[d]	.02	Richter and others 1982
Prescribed underburn	Loblolly pine plantation	Upper Piedmont, South Carolina	.05	.05	Douglass and Van Lear 1983
Clearcut, slash broadcast burned	Douglas-fir, ponderosa pine	Southern Idaho	.02	.05	Clayton and Kennedy 1985
Prescribed burn, moderate	Ponderosa pine, gambel oak	Central Arizona	0.0013[a]	0.0029	Gottfried and DeBano 1990
Wildfire	Chaparral	Lexington Reservoir, Santa Clara County, CA	.02	.04	Taylor and others 1993
Prescribed underburn	White fir, giant sequoia, red fir, sugar pine, Jeffrey pine	Sequoia National Park, CA	.001 – .005	.010 – .394	Chorover and others 1994
Prescribed broadcast:	Chaparral	Southern California			Riggan and others 1994
Moderate burn			2.5	9.5	
Severe burn			2.5	15.3	
Wildfire	Lodgepole pine, Douglas-fir, ponderosa pine, western larch	Glacier National Park, MT	< .040	.124 – .312	Hauer and Spencer 1998

~ = About or approximately.
[a] Maximum level attained.
[b] Mean concentration from May to July after August fire.
[c] Cited in Wright and Watts (1969).
[d] Pretreatment not significantly different from posttreatment.

in vitro nitrite reached concentrations as high as 33.2 ppm. Accidental deposition of retardants in streams has produced values of nitrate and ammonia sufficiently high to be of concern in drinking water.[2] Great caution needs to be exerted to keep retardant chemicals out of streams that are public drinking water sources.

Phosphorus

Phosphate, as a component of fire retardants, can lead to eutrophication. See chapter 3 for discussion of phosphorus impacts on drinking water. Prior to wildfire, phosphate concentrations ranged from 0.007 ppm to 0.17 ppm (Hoffman and Ferreira 1976, Tiedemann and others 1978, Wright and others 1976). After wildfire, prescribed fire, or clearcutting followed by broadcast burning, phosphate concentrations stayed the same or increased only as high as 0.2 ppm (Longstreth and Patten 1975). Any phosphorus added to the stream system may have been taken up by the aquatic organisms and, therefore, little increase was detectable. We found no reports of changes in phosphate concentration as the result of an inadvertent application of retardant directly into a stream.

Sulfur

The sulfate ion is relatively mobile in soil water systems (Johnson and Cole 1977). Although not as well studied as those for nitrogen, the mineralization processes for sulfur are similar. In streamwater from wildland watersheds, observed levels of sulfate are usually low (table 12.6). Control or prefire values range from as low as 1.17 ppm to as high as 66 ppm, while postfire values range from 1.7 ppm to a high of 80.7 ppm, well below the recommended secondary drinking water standard (250 ppm) (table 2.4).

Chloride

Chloride response to fire and clearcutting plus fire has been documented in several studies, and all responses are low (table 12.7). Chloride concentrations in control or prefire samples ranged from 0.49 to 6.4 ppm, and the chloride concentration in postfire samples ranged from 0.40 to 7.1 ppm (Lathrop 1994), well below the recommended secondary drinking water standard (250 ppm) (table 2.4). Lewis Lake in Yellowstone National Park, WY, with its large volume of water, had the highest chloride values for both the prefire and postfire periods among the data examined.

Total Dissolved Solids

Only two studies reported total dissolved solids; many other studies measured some of the constituents of total dissolved solids but not total dissolved solids per se. Hoffman and Ferreira (1976) detected a total dissolved solids concentration of about 11 ppm in the control area and 13 ppm in the burned area, which had been a mixed conifer and shrub stand in Kings Canyon National Park, CA. Lathrop (1994) found Yellowstone Lake in Yellowstone National Park and Lewis Lake had pretreatment total dissolved solids concentrations of 65.8 and 70 ppm. The total dissolved solids concentrations after the fires were 64.8 and 76 ppm, well below the recommended secondary drinking water standard (500 ppm) (table 2.4).

Trace Elements

Fredriksen's (1971) results raise a question about how well we understand the responses of micronutrients or trace elements to fire or to fire after clearcutting. In his stream chemistry profile after clearcutting and broadcast burning, he documented a maximum concentration of manganese of 0.44 ppm, exceeding the recommended secondary drinking water standard (0.05 ppm) (table 2.4), which may raise palatability issues but is not a health risk. There are established drinking water standards for 14 additional trace constituents, including heavy metals. Information on the effects of these elements after a forest or rangeland fire on drinking water quality is lacking.

Effects on Ground Water

Little research has been conducted on the effects of fire, fire suppression, and fire rehabilitation activities on ground water quality. It is reasonable to expect that fire will have little effect on ground water quality. A possible, but unlikely, scenario would be a fire followed by an intense long-duration precipitation event sufficient to cause major flooding, which could contaminate ground water. In such a case, the fire sets the stage for contamination of the ground water source.

[2] Labat-Anderson Incorporated. 1994. Chemicals used in wildland fire suppression: a risk assessment. Prepared for: Fire and Aviation Management, U.S. Department of Agriculture, Forest Service. Contract 53-3187-9-30; Task 93-02. 187 p. Prepared by: Labat-Anderson Incorporated, 2200 Clarendon Boulevard, Suite 900, Arlington, VA 22202.

Table 12.6—Sulfate concentration in water after fire alone

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - *Parts per million* - - - -		
Wildfire	Taiga	Interior Alaska	7.12 – 66	8.3 –80.7	Lotspeich and others 1970
	Mixed conifer, shrub	Central Sierra Nevada Mountains, CA	1.5	1.7	Hoffman and Ferreira 1976
	Pine, spruce, fir, aspen, birch[a]	Northeastern Minnesota lakes	1.17	1.79 – 1.86	Tarapchak and Wright 1977
Prescribed underburn	White fir, giant sequoia, red fir, sugar pine, Jeffrey pine	Sequoia National Park, CA, Log Creek, control	.26	.37, .30, .45[b]	Chorover and others 1994
	White fir, fewer Giant sequoia	Tharp's Creek, burn	.24	9.68, 1.32, 2.15[b]	
Yellowstone wildfires	Subalpine lakes	Yellowstone Lake	8.9	6.4[c]	Lathrop 1994
		Lewis Lake, Yellowstone National Park, WY	4.0	3.0	

[a] Cited in Wright and Watts 1969.
[b] Postburn years one, two, and three, in sequence.
[c] Average of reported median values from four areas of Yellowstone Lake.

Table 12.7—Chloride concentration in water after fire alone

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - *Parts per million* - - - -		
Wildfire	Taiga	Interior Alaska	0.9 – 5.0	1.2 – 4.6	Lotspeich and others 1970
	Mixed conifer, shrub	Central Sierra Nevada Mountains, CA	.6	1.0	Hoffman and Ferreira 1976
	Pine, spruce, fir, aspen, birch[a]	Northeastern Minnesota lakes	.80 – .89	1.24	Tarapchak and Wright 1977
Prescribed underburn	White fir, giant sequoia, red fir, sugar pine, Jeffrey pine	Sequoia National Park, CA	.49 – .56	.40 – 2.78	Chorover and others 1994
Yellowstone wildfires	Subalpine lakes	Yellowstone Lake	5.1	3.6	Lathrop 1994
		Lewis Lake, Yellowstone National Park, WY	6.4	7.1	

[a] Cited in Wright and Watts 1969.

Reliability and Limitations of Findings

The results of research on the effects of fire on drinking water quality are strong and consistent, especially from the Pacific Northwest, the Rocky Mountains, and the Southwest. The results indicate that the effects of fire and fire management practices on water quality are similar within each of these three large areas. Data from the Southeast are somewhat more limited in spite of the region's extensive prescribed fire program. In Alaska, water-quality research after fire has been minimal, even though the area has many wildfires.

With the changes in pH, nitrate, sulfate, and chloride so consistently small, a land manager can safely assume that effects will be similar to those found in the literature, if the treatments and fire severity, slope, and soil and vegetation types are comparable. In areas likely to be nitrogen saturated, such as areas of high soil concentrations of nitrogen from chronic atmospheric deposition, nitrate concentrations in streamwater after a fire may exceed the established maximum contaminant level of 10 ppm (chapter 2, table 2.3). In areas of suspected nitrogen saturation, common sense tells us that nitrogen-containing fertilizer should not be applied. Application of nitrogen fertilizer would exacerbate the risk of degrading source water supplies. See chapter 3 for more discussion of nitrogen saturation.

Results of previous wildland fires can be used as a basis for estimating the effects of new fires on drinking water quality. Fires need to be of the same type; that is, previous wildfires should be used as the comparison basis for new wildfires, and previous prescribed fires as the basis for new prescribed fires. The more factors, such as slope and vegetation, among others, that match between the previously documented fires and the new fires, the closer the approximation. Nevertheless, results and predictions based on limited data must be used cautiously. In two studies (Beschta 1980, Helvey and others 1985), the researchers specifically caution against extrapolating results of turbidity and sediment research beyond the watersheds in which the research was conducted.

Research Needs

1. Research methods need to be carefully selected for measurements of sediment. Suspended sediment concentration is used in some studies while turbidity, the standard measurement, is used in others. At this time, regression relationships between sediment, in parts per million, and turbidity, in NTU's, need to be developed for each individual watershed or stream system. Interpretation of future research results will be facilitated when all measurements are reported in the standard NTU's.

2. Areas with chronic atmospheric deposition, such as those studied by Riggan and others (1994), need further research into the relations between fire and nitrogen release into streams.

3. We have little information on the abundance of trace elements (micronutrients) after fire. When elements, such as lead, copper, fluoride, manganese, iron, zinc, and mercury, among others, are above certain levels, they are important potential contaminants in drinking water supplies. We do not understand the effects of fire in combination with other treatments on micronutrients. Effects may be particularly important for some of the heavy-metal trace elements.

4. The inadvertent application of fire retardants directly into a stream can produce increased levels of nitrate and possibly sulfate, phosphate, and some trace elements. Information is needed about the potential effects of specific retardants on drinking water quality.

5. The BAER practices, particularly the use of contour-felled erosion barriers, need to be systematically studied to determine their effectiveness for reducing storm runoff, erosion, and sediment movement, which pose a risk to the quality of source water for public water supplies.

Key Points

1. When a wildland fire occurs, the principal concerns for change in drinking water quality are: (1) the introduction of sediment; and (2) the potential introduction of nitrates, especially if in areas with chronic atmospheric deposition.

2. As we considered the above types of fire effects on drinking water, several concepts important to the land manager became apparent. The magnitude of the effects of fire on water quality is primarily driven by fire severity, and not necessarily by fire intensity. Fire severity is a qualitative term describing the amount of fuel consumed, while fire intensity is a quantitative measure of the rate of heat release. In other words, the more severe the fire the greater the amount of fuel consumed and nutrients released and the more susceptible the site is to erosion of soil and nutrients into the stream where it could potentially affect water quality. Wildfires usually are more severe than prescribed fires, and, as a result, they are more likely to produce significant effects on water quality. On the other hand, prescribed fires are designed to be less severe and would be expected to produce less effect on water quality. Use of prescribed fire allows the manager the opportunity to control the

severity of the fire and to avoid creating large areas burned at high severity. The degree of fire severity is also related to the vegetation type. For example, in grasslands the differences between prescribed fire and wildfire are probably small. In forested environments, the magnitude of the effects of fire on water quality will probably be much lower after a prescribed fire than after a wildfire because a larger amount of fuel may be consumed in a wildfire. Canopy-consuming wildfires would be expected to be of the most concern to managers because of the loss of canopy coupled with the destruction of soil aggregates. These losses present the worst-case scenario in terms of water quality. The differences between wild and prescribed fire in shrublands are probably intermediate between those seen in grass and forest environments.

3. Another important determinant of the magnitude of the effects of fire on water quality is slope. Steepness of the slope has a significant influence on movement of soil and nutrients into stream channels where it can affect water quality. Wright and others (1976) found that as slope increased in a prescribed fire, erosion from slopes is accelerated. If at all possible, the vegetative canopy on steep, erodible slopes needs to be maintained, particularly if adequate streamside buffer strips do not exist to trap the large amounts of sediment and nutrients than can be transported quickly into the stream channel. It is important to maintain streamside buffer strips whenever possible, especially when developing prescribed fire plans. These buffer strips will capture much of the sediment and nutrients from burned upslope areas.

4. Two more concerns, which are more site-specific, deal with soils. Both the general type of soil and a soil's propensity to develop water repellency can be determinants of the magnitude of the effects of fire on drinking water quality. When sandy soils are burned, nutrient transport and loss are rapid. These soils do not have the ability to capture and hold nutrients, but, rather, allow the nutrients to move into the ground water and eventually into nearby streams. Additionally, in areas with sandy soils, which contain few nutrients, most of the nutrient capital is stored aboveground. A severe fire volatilizes many of these nutrients, impoverishing the site, while adding to the nutrient load in streams. Prescribed fires in these areas need to be very carefully planned to retain as many nutrients on site as possible through the use of low-severity fires.

5. If a site is close to nitrogen saturation, it is possible to exceed maximum contamination levels for drinking water of nitrate (10 ppm) after a severe fire. Such areas should not have nitrogen-containing fertilizer applied after the fire. See chapter 3 for more discussion of nitrogen saturation.

6. The propensity for a site to develop water repellency after fire must be considered. Water-repellent soils do not allow precipitation to penetrate down into the soil and, therefore, are conducive to erosion. Such sites can put large amounts of sediment and nutrients into surface water.

7. Finally, heavy rain on recently burned land can seriously degrade water quality. The effects of the Buffalo Creek wildfire on the water supply for Denver, noted in the first paragraph of this chapter, demonstrated these effects on a water supply. Severe erosion and runoff are not limited to wildfire sites alone. If the storm delivers large amounts of precipitation or is sufficiently intense, accelerated erosion and runoff can occur after a carefully planned prescribed fire. Conversely, if below-average precipitation occurs after a wildfire, there may not be a substantial increase in erosion and runoff.

8. The land manager can influence the effects of fire on drinking water quality by careful prescribed burning. Limiting fire severity, avoiding burning on steep slopes, and limiting burning on sandy or potentially water-repellent soils will reduce the magnitude of the effects of fire on water quality.

Literature Cited

Amaranthus, Michael P. 1989. Effect of grass seeding and fertilizing on surface erosion in two intensely burned sites in southwest Oregon. In: Proceedings of a symposium on fire and watershed management; 1988 October 26–28; Sacramento, CA. Gen. Tech. Rep. PSW–109. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 148–149.

Amaranthus, Michael; Jubas, Howard; Arthus, David. 1989. Stream shading, summer streamflow, and maximum water temperature following intense wildfire in headwater streams. In: Berg, Neil H., tech. coord. Proceedings of a symposium on fire and watershed management; 1988 October 26–28; Sacramento, CA. Gen. Tech. Rep. PSW–109. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 75–78.

American Public Health Association. 1976. Standard methods for the examination of water and wastewater. 14th ed. New York: American Public Health Association, American Water Works Association, and Water Pollution Control Federation. 1,193 p.

Bailey, R.G.; Rice, R.M. 1969. Soil slippage: an indicator of slope instability on chaparral watersheds of southern California. The Professional Geographer. 21: 172–177.

Beschta, R.L. 1980. Turbidity and suspended sediment relationships. In: Symposium on watershed management; 1980 July 21–23; Boise, ID. New York: American Society of Civil Engineers: 271–281.

Beschta, Robert L. 1990. Effects of fire on water quantity and quality. In: Walstad, John D.; Radosevich, Steven R.; Sandberg, David V., eds. Natural and prescribed fire in Pacific Northwest forests. Corvallis, OR: Oregon State University Press: 219–232.

Buhl, Kevin J.; Hamilton, Steven J. 1998. Acute toxicity of fire-retardant and foam-suppressant chemicals to early life states of chinook salmon (*Oncorhynchus tshawytscha*). Environmental Toxicology and Chemistry. 17: 1589–1599.

Campbell, R.E.; Baker, M.B., Jr.; Ffolliott, P.F. [and others]. 1977. Wildfire effects on a ponderosa pine ecosystem: an Arizona case study. Res. Pap. RM–191. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 12 p.

Chorover, Jon; Vitousek, Peter M.; Everson, Daniel A. [and others]. 1994. Solution chemistry profiles of mixed-conifer forests before and after fire. Biogeochemistry. 26: 115–144.

Clayton, James L.; Kennedy, Debora A. 1985. Nutrient losses from timber harvest in the Idaho batholith. Soil Science Society of America Journal. 49: 1041–1049.

Corbett, E.S; Green, L.R. 1965. Emergency revegetation to rehabilitate burned watersheds in southern California. Res. Pap. PSW–22. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 14 p.

Covington, W.W.; Sackett, S.S. 1986. Effect of periodic burning on soil nitrogen concentrations in ponderosa pine. Soil Science Society of America Journal. 50(2): 452–457.

Covington, W. Wallace; Sackett, S.S. 1992. Soil mineral nitrogen changes following prescribed burning in ponderosa pine. Forest Ecology and Management. 54: 175–191.

DeBano, Leonard F.; Neary, Daniel G.; Ffolliott, Peter F. 1998. Fire's effects on ecosystems. New York: John Wiley. 333 p.

Douglass, James E.; Van Lear, David H. 1983. Prescribed burning and water quality in ephemeral streams in the Piedmont of South Carolina. Forest Science. 29(1): 181–189.

Feller, M.C.; Kimmins, J.P. 1984. Effects of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. Water Resources Research. 20(1): 29–40.

Fenn, M.E.; Poth, M.A.; Aber, J.D. [and others]. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecological Applications. 8(3): 706–733.

Fredriksen, R.L. 1971. Comparative chemical water quality—natural and disturbed streams following logging and slash burning. In: Krygier, James T.; Hall, James D., eds. Proceedings of a symposium: forest land uses and stream environments; 1970 October 19–21; Corvallis, OR. Corvallis, OR: Oregon State University: 125–137.

Gaikowski, Mark P.; Hamilton, Steven J.; Buhl, Kevin J. [and others]. 1996. Acute toxicity of three fire-retardant and two fire-suppressant foam formulations to the early life stages of rainbow trout (*Oncorhynchus mykiss*). Environmental Toxicology and Chemistry. 15: 1365–1374.

Gottfried, Gerald J.; DeBano, Leonard F. 1990. Streamflow and water quality responses to preharvest prescribed burning in an undisturbed ponderosa pine watershed. In: Krammes, J.S., tech. coord. Effects of fire management of southwestern natural resources; 1988 November 15–17; Tucson, AZ. Gen. Tech. Rep. RM–191. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 222–228.

Hauer, F. Richard; Spencer, Craig N. 1998. Phosphorus and nitrogen dynamics in streams associated with wildfire: a study of immediate and long-term effects. International Journal of Wildland Fire. 8(4): 183–198.

Helvey, J.D. 1972. First-year effects of wildfire on water yield and stream temperature in north-central Washington. In: Csallany, Sandor C.; McLaughlin, Thad G.; Striffler, William D., eds. Watersheds in transition; 1972 June 19–22; Fort Collins, CO. Urbana, IL: American Water Resources Association: 308–312. Vol. 14.

Helvey, J.D. 1980. Effects of a north central Washington wildfire on runoff and sediment production. Water Resources Bulletin. 16(4): 627–634.

Helvey, J.D.; Tiedemann, A.R.; Anderson, T.D. 1985. Plant nutrient losses by soil erosion and mass movement after wildfire. Journal of Soil and Water Conservation. 40(1): 168–173.

Hoffman, Ray J.; Ferreira, Rodger F. 1976. A reconnaissance of the effects of a forest fire on water quality in Kings Canyon National Park, California. Open-File Rep. 76–497. Menlo Park, CA: U.S. Department of the Interior, U.S. Geological Survey. 17 p.

Illg, Cathy; Illg, Gordon. 1997. Forest fire, water quality, and the incident at Buffalo Creek. American Forests: (Winter/Spring): 33–35.

Johnson, D.W.; Cole, D.W. 1977. Anion mobility in soils, relevance to nutrient transport from terrestrial to aquatic systems. EPA–600/3–77–068. Corvallis, OR: U.S. Environmental Protection Agency, Environmental Research Laboratory. 27 p.

Klock, G.O.; Tiedemann, A.R.; Lopushinsky, W. 1975. Seeding recommendations for disturbed mountain slopes in north central Washington. Res. Note PNW–244. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 8 p.

Lathrop, Richard G., Jr. 1994. Impacts of the 1988 wildfires on the water quality of Yellowstone and Lewis Lakes, Wyoming. International Journal of Wildland Fire. 4(3): 169–175.

Levno, Al; Rothacher, Jack. 1969. Increases in maximum stream temperatures after slash burning in a small experimental watershed. Res. Note PNW–110. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 7 p.

Longstreth, D.J.; Patten, D.T. 1975. Conversion of chaparral to grass in central Arizona: effects on selection ions in watershed runoff. American Midland Naturalist. 93(1): 25–34.

Lotspeich, Frederick B.; Mueller, Ernst W.; Frey, Paul J. 1970. Effects of large scale forest fires on water quality in interior Alaska. College, AK: U.S. Department of the Interior, Federal Water Pollution Control Administration, Northwest Region, Alaska Water Laboratory. 115 p.

McDonald, Susan F.; Hamilton, Steven J.; Buhl, Kevin J.; Heisinger, James F. 1996. Acute toxicity of fire control chemicals to *Daphnia magna* (Straus) and *Selenastrum capricornutum* (Pintz). Ecotoxicology and Environmental Safety. 33: 62–72.

McDonald, Susan F.; Hamilton, Steven J.; Buhl, Kevin J.; Heisinger, James F. 1997. Acute toxicity of fire control chemicals to *Hyalella azt*eca (Saussure). Environmental Toxicology and Chemistry. 16: 1370–1376.

Monleon, V.J.; Cromack, K., Jr.; Landsberg, J.D. 1997. Short- and long-term effects of prescribed underburning on nitrogen availability in ponderosa pine stands in central Oregon. Canadian Journal of Forest Research. 27(3): 369–378.

Norris, L.A.; Hawkes, C.L.; Webb, W.L. [and others]. 1978. A report of research on the behavior and impact of chemical fire retardants in forest streams. Corvallis, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Forestry Sciences Laboratory. 151 p.

Norris, Logan; Webb, Warren L. 1989. Effects of fire retardant on water quality. In: Berg, Neil H., tech. coord. Proceedings of the symposium on fire and watershed management; 1988 October 26–28; Sacramento, CA. Gen. Tech. Rep. PSW–109. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 78–86.

Pase, C.P.; Ingebo, P.A. 1965. Burned chaparral to grass: early effects on water and sediment yields from two granitic soil watersheds in Arizona. In: Proceedings of the Arizona watershed symposium; 1965 September 22; Tempe, AZ. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 8–11.

Raison, R.J.; Khanna, P.K.; Woods, P.V. 1985. Mechanisms of element transfer to the atmosphere during vegetation fires. Canadian Journal of Forest Research. 15: 132–140.

Rich, L.R. 1962. Erosion and sediment movement following a wildfire in a ponderosa pine forest of central Arizona. Res. Note RM–168. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 12 p.

Richter, D.D.; Ralston, C.W.; Harms, W.R. 1982. Prescribed fire: effects on water quality and forest nutrient cycling. Science. 215: 661–663.

Riggan, Philip J.; Lockwood, Robert N.; Jacks, Paula M. [and others]. 1994. Effects of fire severity on nitrate mobilization in watersheds subject to chronic atmospheric deposition. Environmental Science and Technology. 28(3): 369–375.

Robichaud, P.R.; Beyers, J.L.; Neary, D.G. 2000. Evaluating the effectiveness of post-fire rehabilitation treatments. Gen. Tech. Rep. RMRS–63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 85 p.

Robichaud, P.R.; Waldrop, T.A. 1994. A comparison of surface runoff and sediment yields from low- and high-severity site preparation burns. Water Resources Bulletin. 30(1): 27–34.

Sims, B.D.; Lehman, G.S.; Ffolliott, P.F. 1981. Some effects of controlled burning on surface water quality. In: Hydrology and water resources in Arizona and the southwest; Proceedings of the 1981 meetings of the Arizona section—American Water Resources Association and the hydrology section—Arizona Academy of Science; 1981 May 1–2; Tucson, AZ. [Place of publication unknown]: Arizona Section—American Water Resources Association and the Hydrology Section—Arizona Academy of Science. 11: 87–90.

Society of American Foresters. 1995. Reauthorization of the Clean Water Act: forestry effects on water quality. SAF 95–01. Bethesda, MD: Society of American Foresters. 24 p.

Spencer, Craig N.; Hauer, F. Richard. 1991. Phosphorus and nitrogen dynamics in streams during a wildfire. The North American Benthological Society. 10(1): 24–30.

Swift, L.W.; Messer, J.B. 1971. Forest cuttings raise temperature of small streams in the Southern Appalachians. Journal of Soil and Water Conservation. 26(3): 111–116.

Tarapchak, Stephen J.; Wright, Richard F. 1977. Three oligotrophic lakes in northern Minnesota. In: Seyb, L.; Randolph, K., eds. North American project—a study of U.S. water bodies. Corvallis, OR: U.S. Environmental Protection Agency: 64–90.

Taylor, Marcus J.; Shay, Johnevan M.; Hamlin, Scott N. 1993. Changes in water quality conditions in Lexington Reservoir Santa Clara County California following a large fire in 1985 and flood in 1986. Water-Res. Invest. Rep. 92–4172. Denver: U.S. Department of the Interior, U.S. Geological Survey. 23 p.

Tiedemann, A.R. 1987. Combustion losses of sulfur from forest foliage and litter. Forest Science. 33(1): 216–223.

Tiedemann, A.R.; Driver, C.H. 1983. *Snow eriogonum*: a native halfshrub to revegetate winter game ranges. Reclamation and Revegetation Research. 2: 31–39.

Tiedemann, A.R.; Helvey, J.D.; Anderson, T.D. 1978. Stream chemistry and watershed nutrient economy following wildfire and fertilization in eastern Washington. Journal of Environmental Quality. 7(4): 580–588.

Tiedemann, Arthur R. 1973. Stream chemistry following a forest fire and urea fertilization in north-central Washington. Res. Note PNW–203. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 20 p.

Tiedemann, Arthur R. 1981. Stream chemistry, nutrient economy, and site productivity consequences of wildland management and wildfire. In: Baumgartner, David M., ed. and comp. Interior west watershed management; 1980 April 8–10; Spokane, WA. Pullman, WA: Washington State University, Cooperative Extension Service: 183–201.

Tiedemann, Arthur R.; Conrad, Carol E.; Dieterich, John H. [and others]. 1979. Effects of fire on water: a state of knowledge review. In: National fire effects workshop; 1978 April 10–14; Denver. Gen. Tech. Rep. WO–10. Washington, DC: U.S. Department of Agriculture, Forest Service. 29 p.

Tiedemann, Arthur R.; Klock, Glen O. 1976. Development of vegetation after fire, reseeding, and fertilization on the Entiat Experimental Forest. In: Annual proceedings of the Tall Timbers fire ecology conference; 1974 October 16–17; Missoula, MT. Tallahassee, FL: Tall Timbers Research Station: 171–192.

Ursic, S.J. 1969. Hydrologic effects of prescribed burning on abandoned fields in northern Mississippi. Res. Pap. SO–46. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 20 p.

Ursic, S.J. 1970. Hydrologic effects of prescribed burning and deadening upland hardwoods in northern Mississippi. Res. Pap. SO–54. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 15 p.

U.S. EPA. 1999. Current drinking water standards, national primary and secondary drinking water regulations. U.S. Environmental Protection Agency, Office of Ground Water and Drinking Water. [Available on the Web at: http://www.epa.gov/safewater/mcl.html].

Vitousek, Peter M.; Melillo, Jerry M. 1979. Nitrate losses from disturbed forests: patterns and mechanisms. Forest Science. 25(4): 605–619.

Wells, Carol G.; Campbell, Ralph E.; DeBano, Leonard F. [and others]. 1979. Effects of fire on soil: a state of knowledge review. In: Forest Service national fire effects workshop; 1978 April 10–14; Denver. Gen. Tech. Rep. WO–7. Washington, DC: U.S. Department of Agriculture, Forest Service. 34 p.

Wells, Wade G., II. 1986. Fire dominates sediment production in California chaparral. In: Dell, B.; Hopkins, A.J.M.; Lamont, B.B., eds. Resilience in Mediterranean-type ecosystems; 1984 August; Perth, Western Australia. Boston: Kluwer Academic Publishers: 163–164.

Wright, H.E., Jr.; Watts, W.A. 1969. Glacial and vegetational history of northeastern Minnesota. Spec. Publ. SP–11. Minneapolis, MN: University of Minnesota, Minnesota Geological Survey. 59 p.

Wright, Henry A.; Churchill, Francis M.; Stevens, W. Clark. 1976. Effect of prescribed burning on sediment, water yield, and water quality from dozed juniper lands in central Texas. Journal of Range Management. 29(4): 294–298.

Wright, Henry A.; Churchill, Francis M.; Stevens, W. Clark. 1982. Soil loss, runoff, and water quality of seeded and unseeded steep watersheds following prescribed burning. Journal of Range Management. 35(3): 382–385.

Chapter 13

Pesticides

J.L. Michael[1]

Introduction

On forest and grassland, management often must protect desirable vegetation from pathogens, competing vegetation, insects, and animals. Vegetation also is managed to clear road and utility rights-of-way, to improve recreation areas and wildlife habitat, and to control noxious weeds. Pesticides offer inexpensive and effective ways of getting these jobs done. The Forest Service requires: (1) training of personnel who recommend and use pesticides, (2) applicator certification, and (3) safety plans to assure the safety of personnel and the protection of environmental values like drinking water quality. Nonnational forest land is treated with pesticides for many of the same purposes, but often more intensively.

The Federal Insecticide, Fungicide, and Rodenticide Act, as amended (FIFRA) (Public Law 92–516, and 40 CFR 158) allows the registration of pesticides for use in the United States. The registration process is an extraordinary one that requires years of testing before sufficient efficacy, environmental safety, toxicology, and public safety data can be collected and evaluated in the support of registration of a new pesticide. While this process is designed to assure safety, new and old pesticides, following registration, continue to be studied by researchers in private, State, and Federal agencies in an effort to identify any potential environmental or toxicological problems. An integral part of protecting public health and environmental values during pesticide use is the requirement that they must be applied according to directions approved by the U.S. Environmental Protection Agency (EPA) and included on the label of every registered pesticide. Under FIFRA, pesticide labels are legally binding documents, and any infraction of the directions for application is a violation of law. Users of pesticides must exercise extreme caution in following label directions and must also exercise good judgement, especially when pesticide use is planned in an area near municipal water supplies. In addition, pesticide users must provide adequate handling facilities for mixing and storage and be well prepared to deal with spills.

[1] Research Ecologist, USDA Forest Service, Southern Research Station, Auburn, AL.

To meet the minimum requirements of FIFRA at the State level, the EPA has established and maintains cooperative enforcement agreements for pesticide use inspections, producer establishment inspections, marketplace surveillance, applicator certification, and experimental use inspections. State government is responsible for (1) certification of pesticide applicators, (2) enforcement of FIFRA pesticide use regulations and inspections, (3) endangered species considerations, (4) worker protection, and (5) ground water protection.

When forestry pesticides are used near water on Federal, State, or privately owned land, buffer zones are left between the treated areas and the water resource (see chapter 5). The width of the buffer varies with site conditions, site sensitivity, and local or State recommendations. National forests in some States use more conservative buffers than those recommended by the State. Comerford and others (1992) have reviewed many agricultural studies in an attempt to draw inferences regarding effectiveness of buffer strips in mediating stream contamination. However, relatively little research data are available on effectiveness of buffers on forest sites. It, therefore, is not possible to determine the minimum buffer width to protect streams from either pesticide or sediment contamination.

Issues and Risks, Pesticide Application

Approximately 16 percent of the 3.6 million square miles of land in the United States is treated with pesticides annually (Pimentel and Levitan 1986). The most intensive use of pesticides occurs on land occupied by households. Household tracts account for only 0.4 percent of all land but receive 12 percent of all pesticides used in the United States. Agricultural land (52 percent of all land) is the next most intensively treated, receiving 75 percent of all pesticides used. Government and industrial land (16 percent of all land) receives 12 percent of all pesticides. The least intensive use of pesticides occurs on forest land (32 percent of the land). Pimentel and Levitan (1986) point out that forest land receives only 1 percent of all pesticides used in the United States and that <1 percent of all forest land is treated annually.

A wide variety of pesticides are used on forest and grasslands. Table 13.1 lists these pesticides and the purposes for which they are used. Maximum Contaminant Levels (MCL's), established by EPA, are listed in chapter 2, table 2.3.

Biological control agents, including *Bacillus thuringiensis* (Bt) and nucleopolyhedrosis virus (Npv), are used for control of western spruce budworms and gypsy moths. While these are pathogens of insects, they have no known impacts on drinking water quality.

Plant pathogens represent a potential problem throughout forest and rangeland ecosystems, but their destructive impacts are most severe in seedling nurseries and seed storage facilities. Fungicides and fumigants are used to control these pathogens on seeds, in seedling nurseries, in greenhouses, in seed tree nurseries, and on individual trees. In 1997, the Forest Service treated 35 greenhouses with fungicides and fumigants including benomyl, chlorothalonil, dicloran, iprodione, metalaxyl, propiconazole, thiophanate methyl, and triadimefon. Most fungicide and fumigant use occurs on small acreages in nurseries for disease control.

Small amounts of strychnine and putrescent egg solids were used over extensive acreages for animal damage control. While very small amounts of strychnine were used over vast acreages, it is very toxic (table 13.2). Putrescent egg solids, by comparison, are derived from food products and the EPA has waived toxicology requirements. These two products accounted for more than 96 percent of the active ingredients used in protection of vegetation from animals. Insect control relied mainly on biological agents, but some insecticides and oils were used. The insecticides included carbaryl and chlorpyrifos. Dormant oil was used for control of a variety of insects and their eggs. These three (carbaryl, chlorpyrifos, and dormant oil) represent 94.6 percent of all chemical insecticides used on National Forest System (NFS) land.

Vegetation management is frequently taken to mean the control of competing vegetation in timber management programs. On NFS land, more than three times as much land was treated for protection of vegetation from animals and insects and to control noxious weeds, than for control of competing vegetation in timber management programs in fiscal year 1997 (U.S. Department of Agriculture, Forest Service 1998). Competing vegetation can be controlled with herbicides, algicides, and plant growth regulators. Table 13.1 shows management objectives for herbicide use on NFS land in fiscal year 1997. Many acres are treated for timber management, principally planting site preparation and release of crop trees. Such treatments usually occur only once or twice over a rotation. Rotation length depends on tree species, site productivity, and management objectives. The rotation may be as short as 20, or longer than 150 years. Thus, herbicides are used in timber management only once or twice in 20 or more years. Treatments were for site preparation, conifer release, and hardwood release.

Noxious weed control is often accomplished by treatment with herbicides. Noxious weeds are usually nonnative plants that, lacking natural controls, spread quickly and take over or ruin habitat for native plants. They generally possess one or more of the following characteristics: aggressive and difficult to manage, poisonous, toxic, parasitic, and a carrier or host of serious insects or disease. There are 74 terrestrial species on the Federal noxious weed list, including kudzu. The frequency of noxious-weed treatment varies by species. In southern forests, kudzu requires annual treatment over several years for effective control. Typically, attempts to control noxious weeds do not eradicate them, but bring them under enough control to reduce immediate problems. Timber management (45.2 percent) and noxious weed control (48.8 percent) accounted for 94 percent of all acres treated with herbicides.

Protecting forests and seedlings from animal pests is the single largest component of the vegetation management program. Rabbits and deer were the most common target mammals, while western spruce budworms and gypsy moths were the principal target insects.

One major issue with pesticide use is the impact on drinking water quality. To adversely impact drinking water, pesticides must (1) be harmful to humans, and (2) reach drinking water at concentrations exceeding toxic levels for humans.

Issues and Risks, Toxicity

The toxicity of a chemical is a measure of its ability to harm individuals of the species under consideration. This harm may come from interference with biochemical processes, interruption of enzyme function, or organ damage. Toxicity may be expressed in many ways. Probably the best known term is LD_{50}, the dose at which 50 percent of the test animals are killed. More useful terms have come into popular usage in the last decade: no observed effect level (NOEL), no observed adverse effect level (NOAEL), lowest observed adverse effect level (LOAEL), reference dose (RfD), and, relating specifically to water, the health advisory level (HA or HAL). The EPA uses these terms extensively in risk assessment programs to indicate levels of exposure deemed safe for humans, including sensitive individuals. They are derived from toxicological test data and have built-in safety factors ranging upward from 10, depending on EPA's evaluation of the reliability of the test data.

Table 13.1—Management uses of pesticides commonly used on national forests

Pesticide	Vegetation management use(s)
2, 4-D	Housekeeping and facilities maintenance, noxious weed control, nursery weed control, recreation improvement, right-of-way vegetation management, seed orchard protection agricultural weed control, other vegetation management
Bacillus thuringiensis	Insect suppression
Borax	Disease control
Carbaryl	Insect suppression in the field and in greenhouses, nursery insect control
Chloropicrin	Nursery disease control
Chlorpyrifos	Housekeeping and facilities maintenance, insect control, nursery insect control
Clopyralid	Housekeeping and facilities maintenance, noxious weed control, nursery weed control, right-of-way vegetation management, wildlife habitat improvement
Dazomet	Fungus control, nursery disease control, soil fumigation
Dicamba	Noxious weed control, other vegetation management
Dormant oil	Insect control
Hexazinone	Wildlife habitat improvement, site preparation, conifer release
Imazapyr	Conifer release, hardwood release, hardwood control, noxious weed control, site preparation
Methyl bromide	Nursery disease control, soil fumigation
Metsulfuron	Noxious weed control
Nucleopolyhedrosis virus	Insect suppression
Picloram	Noxious weed control, right-of-way vegetation management, weed control, wildlife habitat improvement
Putrescent egg solids	Animal damage control
Strychnine	Animal damage control, seed orchard protection
Thiram	Animal damage control, fungus control, nursery disease control
Triclopyr	Conifer release, hardwood control, hardwood release, noxious weed control, recreation improvement, right-of-way vegetation management, seed orchard protection, site preparation, thinning, general weed control, wildlife habitat improvement
Zinc phosphide	Animal damage control

The NOEL is determined from animal studies in which a range of doses is given daily; some doses cause adverse effects and others do not (U.S. EPA 1993). The NOAEL is derived from the test data where all doses have some effect, but some of the observed effects are not considered adverse to health. When EPA has data from a number of these tests, the lowest NOEL or NOAEL is divided by a safety factor of at least 100 to determine the RfD. The RfD is an estimate of a daily exposure to humans that is likely to be without an appreciable risk of deleterious effects during a lifetime.

Drinking water standards are calculated for humans by assuming that an adult weighs 155 pounds and consumes 2 pints of water per day, and a child weighs 22 pounds and consumes 1 pint of water per day over the period of exposure. The HAL's are calculated for 1 day, 10 days, longer term (10 percent of life expectancy), or lifetimes (70 years) by dividing the NOAEL or LOAEL by a safety factor and multiplying the resulting value by the ratio of body weight to amount of water consumed daily (U.S. EPA 1993). The safety factor can range from as low as 1, but is rarely <10, and goes as high as 10,000, depending on the available

Table 13.2—Estimates of safe levels for daily exposure to the 20 pesticides most used on National Forest System lands in fiscal year 1997 in the vegetation management program

Pesticide	RfD	NOEL	NOAEL	Lifetime HAL	Reference
	- - - - - *Milligrams per kilogram* - - - - -			*mg/L*	
Borax	0.09	NA	8.8	0.60[a]	U.S. EPA 1990
Carbaryl	.1	NA	9.6	.700	U.S. EPA 1989
Chloropicrin[b]	NA	NA	NA	NA	
Clopyralid	NA	NA	NA	NA	
Chlorpyrifos	.003	0.03	NA	.020	U.S. EPA 1993
2,4-D[c]	.01	NA	1	.070	U.S. EPA 1989
Dazomet[b]	NA	NA	NA	NA	
Dicamba	.03	NA	3	.200	U.S. EPA 1989
Dormant oil	NA	NA	NA	NA	
Glyphosate[c]	.1	20	NA	.700	U.S. EPA 1989
Hexazinone	.05	5	NA	.400	U.S. EPA 1996
Imazapyr	NA	250	NA	NA	U.S. EPA 1997
Methyl bromide	.0014	NA	1.4	.010	U.S. EPA 1990
Metsulfuron	.25	25	NA	NA	U.S. EPA 1988b
Picloram[c]	.007	7	NA	.500	U.S. EPA 1988a
Putrescent egg solids	NA[d]	NA	NA	NA	
Strychnine	.0003	None	None	NA	U.S. EPA 1998a
Thiram	.005	5	NA	NA	U.S. EPA 1992
Triclopyr	.05	5	NA	NA	U.S. EPA 1998b
Zinc phosphide	.0003	None	None	NA	U.S. EPA 1998a

HAL = health advisory level; NA = not available; NOAEL = no observed adverse effect level; NOEL = no observed effect level; RfD = reference dose.
[a] HAL for elemental boron.
[b] These fumigants are not expected to get into water.
[c] Maximum contaminant levels for glyphosate (0.700 mg per liter), 2,4-D (0.070 mg per liter), and picloram (0.500 mg per liter) are discussed in chapter 2.
[d] Made from food products; toxicology was waived by U.S. Environmental Protection Agency.

toxicological data. A safety factor of 10 is used when good NOAEL data are based on human exposures and are supported by chronic or subchronic data in other species. When NOAEL's are available for one or more animal species but not humans and good data for LOAEL in humans is available, a safety factor of 100 is used. When good chronic data are available identifying an LOAEL but not an NOAEL for one or more animal species, a safety factor of 1,000 is used. For situations where good chronic data are absent, but subchronic data identify an LOAEL but not an NOAEL, the safety factor of 10,000 is used. The EPA's estimates of safe levels for daily exposure to the pesticides most widely used on NFS land are summarized in table 13.2. Of the pesticides listed in table 13.2, elemental boron (potentially from borax) and methyl bromide are listed in EPA's drinking water contaminant candidate list for consideration for possible regulation. The MCL's have been established for 2,4-D [0.070 milligrams (mg) per liter], glyphosate (0.700 mg per liter), and picloram (0.500 mg per liter) and these are the same as the already established lifetime HAL's (table 13.2). Information on specific pesticides can be retrieved from the National Pesticides Telecommunication Network at http://ace.orst.edu/info/nptn, EPA site at http://www.epa.gov/epahome/search.html, Extension Toxicology Network at http://ace.orst.edu/info/extoxnet, Material Data Safety Sheets at http://siri.uvm.edu/msds, USDA Forest Service at http://www.fs.fed.us/foresthealth/pesticide, and many others.

Findings from Studies

Pesticides used by the NFS in vegetation management are used around the World in agricultural, forest, range, and urban applications. Some have been found in surface water, shallow ground water, and even in shallow wells (<30 ft), but in concentrations far below levels harmful to human health, and the occurrence is infrequent. Table 13.3 summarizes reports of pesticides from table 13.2 that have been detected in water in the United States.

Larson and others (1997) summarized the results of 236 studies throughout the United States on pesticide contamination of surface water by listing the maximum observed concentrations from each study. These studies were located principally around large river drainage basins and, therefore, represent cumulative pesticide contributions from a wide variety of uses. Monitoring results were reported for 52 pesticides approved for agricultural, urban, and forestry use and their metabolic byproducts. Of the pesticides listed in table 13.2, only six were reported to be present in surface water by Larson and others (1997). They were carbaryl, 1 report; hexazinone, 1 report; chlorpyrifos, 3 reports; picloram, 4 reports; dicamba, 5 reports; and 2,4-D, 24 reports. None of the reported concentrations exceeded EPA safe levels for human health except where application included placement directly in stream channels and most were <0.002 mg per liter. It is important to recognize that surface water is not necessarily drinking water. The studies summarized by Larson and others (1997) dealt with surface water, principally in lakes, reservoirs, and rivers, which would be treated prior to use for drinking. Thus, use of these pesticides according to label directions has not resulted in impairment of drinking water.

Reports of pesticide contamination of water are usually from agricultural (Kolpin and others 1997, Koterba and others 1993) or urban applications (Bruce and McMahon 1996), but the potential exists for contamination from forest vegetation management. Water from forests is generally much less contaminated than water from other land uses. Several studies on forest sites listed in table 13.3 present data for water collected directly from treated areas. The concentration of pesticides can appear quite high compared to samples taken from large rivers and lakes. Pesticide concentrations are greatly reduced by dilution as they move from the treated sites to downstream locations. Degradation of pesticides by biological, hydrolytic, and photolytic routes also contributes to downstream reductions in pesticide concentrations.

From 1985–87, Cavalier and others (1989) monitored 119 wells, springs, and municipal water supplies for occurrence of pesticides in drinking water throughout the State of Arkansas. Monitored wells were generally located in the eastern portion of Arkansas, but eight wells were located in the Ouachita National Forest. Only sites considered highly susceptible to contamination from pesticide use were monitored, and these included domestic, municipal, and irrigation wells. Detection limits for the three forestry pesticides monitored (2,4-D, hexazinone, and picloram) ranged from 70 to 800 times lower than their HAL's. They did not detect well water contamination from any of the 18 pesticides for which they monitored. Failure to detect pesticides in these wells believed to be at high risk for contamination is a very strong indicator that ground water is not at risk from forestry pesticides used according to label directions.

Michael and Neary (1993) reported on 23 studies conducted on industrial forests in the South in which whole watersheds received herbicide treatment. Water flowing from the sites was sampled near the downstream edge of the treatments. The watersheds were relatively small (<300 acres) and the ephemeral to first-order streams draining these watersheds were too small to be public drinking water sources, but their flow reached downstream reservoirs. The maximum observed hexazinone, imazapyr, picloram, and sulfometuron concentrations in streams on these treated sites did not exceed HAL's, except for one case in which hexazinone was experimentally applied directly to the stream channel. Even in this case in which hexazinone was applied directly to the stream at a very high rate, drinking water standards were exceeded for only a few hours. In another study, picloram was accidentally applied directly to streams, but maximum picloram concentrations did not exceed HAL's during the year after application.

Bush and others (1990) reported on use of hexazinone on two Coastal Plain sites (deep sand and sandy loam soils) that were monitored for impacts on ground water. Hexazinone was not detected in ground water at the South Carolina site for 2 years after application. In Florida, hexazinone was found infrequently in shallow test wells at concentrations up to 0.035 mg per liter, much lower than the safe levels for daily exposure (0.400 mg per liter). Water from these sites drains into other creeks and rivers and is diluted before entering reservoirs.

Michael and others (1999) reported the dilution of hexazinone downstream of treated sites. One mile below the treated site, hexazinone concentrations were diluted to one-third to one-fifth the concentration observed on the treated site. Hexazinone was applied for site preparation at 6 pounds active ingredient (ai) per acre to clay loam soils, a rate three times the normal, and it was applied directly to

Table 13.3—Frequency and occurrence of surface and ground water contamination from pesticide use in North America[a]

Pesticide	Water type	Location	Maximum	Range	Comments	Reference
			- - - - - *Milligrams per liter* - - - - -			
2, 4-D	S	Large river basins throughout the United States	0.0075	0.00004 – 0.0075	Twenty-four reports of mainly urban, suburban, agricultural sources	Larson and others 1997
	S	Streams in Oregon and California	2.0	ND – 2.0	Highest concentrations observed from forest areas where no attempt was made to prevent application to water	Norris 1981
	G	Saskatchewan, Canada	.0000007	NG	Natural spring flow	Wood and Anthony 1997
	G	Connecticut, Iowa, Kansas, Maine, Mississippi, South Dakota	.049	.0002 – .049	Well water samples, except for South Dakota, from shallow sand and gravel aquifer	Funari and others 1995
Borax		NR	NR	NR	NR	NR
Carbaryl	S	Mississippi River	.0001	NG	One report	Larson and others 1997
	S	New Brunswick, Canada	.314	NG	Aerial spray spruce budworm control	Sundaram and Szeto 1987
	S	New Brunswick, Canada	.314	.123 – .314	Budworm control	Holmes and others 1981
Chloropicrin		NR	NR	NR	NR	NR
Chlorpyrifos	S	Mississippi River, the lower Colorado River, rivers and lakes in Kansas, irrigation ditches in California Arizona, Nevada	.00015	.00004 – .00015	Three reports	Larson and others 1997
Clopyralid		NR	NR	NR	NR	NR
Dazomet		NR	NR	NR	NR	NR
Dicamba	S	USFS land near Hebo, OR	.037	.006 – .037	Treated 166 ac of 603-ac forest catchment; highest concentration diluted to 0.006 mg/L 2.2 mi downstream	Norris 1975
Glyphosate	S	45-ha coastal British Columbian catchment	.162	.0032 – .162	Highest concentration in streams intentionally sprayed, lowest in streams with smz	Feng and others 1990
	S	Quebec, Canada	3.080	.078 – 3.08	Nine of 36 streams contained glyphosate after forest spraying	Leveille and others 1993
	S	Ohio	5.2	NG	No-tillage establishment of fescue	Edwards and others 1980
	S	Georgia Michigan Oregon	.035 1.237 .031	NG NG NG	Forest sites for scrub-hardwood control and direct spray of streams	Newton and others 1994

continued

Table 13.3—Frequency and occurrence of surface and ground water contamination from pesticide use in North America[a] (continued)

Pesticide	Water type	Location	Maximum	Range	Comments	Reference
			- - - - - *Milligrams per liter* - - - - -			
Glyphosate (cont.)	G	Newfoundland, Canada	0.045	0.004 – 0.045	Application of 4 lbs ai/ac to power substations resulted in contamination of water in monitoring wells	Smith and others 1996
Hexazinone	S	Mississippi River	.00007	NG	Detected in five tributaries	Larson and others 1997
	S	Alabama, Florida, Georgia	.037	.0013 – 0.037	Seven reports, each treated catchment containing ephemeral/first-order streams	Michael and Neary 1993
	S	Alabama	2.400	NG	Applied directly to ephemeral channel and in first runoff water	Miller and Bace 1980
	S	Alabama	.473	.422 – .473	Ephemeral/first-order stream in catchments treated with 3x rate of hexazinone in liquid and pellet formulation with accidental application to streams	Michael and others 1999
	S	Arkansas	.014	NG	11.5-ha watershed drained by ephemeral to first-order stream	Bouchard and others 1985
	S	Georgia	.442	NG	Ephemeral/first-order stream in treated catchment, pellets applied to stream channel	Neary and others 1986
	G	NG[b]	.009	NG	Only one value reported from a single study	Funari and others 1995
Imazapyr	S	Alabama	.680	.130 – .680	Two reports, each treated catchment containing ephemeral/first-order streams, herbicide accidentally applied to stream channel	Michael and Neary 1993
Methyl bromide		NR	NR	NR	NR	NR
Metsulfuron	S G	Central Florida	.008 .002	NG NG	Water in surface depression in slash pine site and 1 of 207 shallow (6-ft) well samples	Michael and others 1991
Picloram	S	North-central Arizona	.32	NG	Pinyon-juniper site	Johnsen 1980
	S	Streams and rivers in North Dakota, Wyoming, Montana	.005	.00001 – .005	Four reports from mainly rangeland uses	Larson and others 1997
	S	Alabama	.442	NG	Pellets accidentally applied directly to forest stream	Michael and others 1989
	S	Georgia, Kentucky, Tennessee	.021	ND – .021	Six study catchments with ephemeral/first-order stream in each treated forest catchment	Michael and Neary 1993
	S	North Carolina	.01	NG	Ephemeral/first-order stream in treated forest catchment	Neary and others 1985

continued

Table 13.3—Frequency and occurrence of surface and ground water contamination from pesticide use in North America[a] (continued)

Pesticide	Water type	Location	Maximum	Range	Comments	Reference
			- - - - - *Milligrams per liter* - - - - -			
Picloram (cont.)	G	Saskatchewan, Canada	0.000225	NR	Natural spring flow	Wood and Anthony 1997
	G	Iowa, Maine, Minnesota, North Dakota	.049	0.00063 – 0.049	Fewer than 2% of well samples were positive	Funari and others 1995
Strychnine		NR	NR	NR	NR	NR
Thiram		NR	NR	NR	NR	NR
Triclopyr	S	Florida	.002	NG	Coastal Plain flatwoods catchments near Gainesville, FL	Bush and others 1988
	S	Ontario, Canada	.35	.23 – .35	Intentional aerial application to boreal forest stream	Thompson and others 1991
Zinc phosphide		NR	NR	NR	NR	NR

Ai = active ingredient; G = ground water; ND = not detected; NG = not given; NR = no reports found in published literature; S = surface water; smz = streamside management zone; USFS = USDA Forest Service.
[a] This table summarizes the levels of pesticides reported in the literature at specific sites and is representative of the literature from North America. However, it cannot be extrapolated for purposes of prediction.
[b] The authors do not provide specific location.

a stream segment, resulting in a maximum observed on-site concentration of 0.473 mg per liter. This was slightly more than the lifetime HAL but considerably below the longer term HAL of 9.0 mg per liter (U.S. EPA 1990). Following the application, on-site stream concentrations did not exceed the lifetime HAL.

Norris (1975) reported contamination of streamflow with dicamba used for control of hardwoods on silty clay loam soils in Oregon. On a 603-acre watershed, 166 acres were aerially sprayed with 1 pound ai per acre [1.1 kilograms (kg) per hectare] of dicamba. A small stream segment was also sprayed causing detectable dicamba residues 2 hours after application began, approximately 0.8 miles (1.3 kilometers) downstream. Concentrations rose for approximately 5.2 hours after treatment began and reached a maximum concentration of 0.037 mg per liter, less than one-fifth of the HAL (0.200 mg per liter). No dicamba residues were detected beyond 11 days after treatment.

Glyphosate and 2,4-D have aquatic labels, which permit direct application to water. Stanley and others (1974) found that when 2,4-D was applied to reservoirs for aquatic weed control, about half of water samples from within treatment areas contained 2,4-D, and the highest concentration (0.027 mg per liter) was less than half of the HAL (0.070 mg per liter). Newton and others (1994) aerially applied glyphosate at three times the normal forestry usage rate [4 pounds ai per acre (4.4 kg per hectare)], no buffers were left, and all streams and ponds were sprayed. Initial water concentrations were 0.031 and 0.035 mg per liter in Oregon and Georgia, respectively, and 1.237 mg per liter in Michigan on the day of application. After day one, glyphosate concentrations dropped to below 0.008 mg per liter on all three sites for the duration of the study. The HAL was exceeded on only one of three sites and then for only 1 day.

There is little information on the movement of metsulfuron to streams. Michael and others (1991) found trace residues of metsulfuron in shallow monitoring wells in Florida where 24 wells were sampled to a depth of 6 feet (1.8 meters). Metsulfuron was detected (0.002 mg per liter) in 1 of 207 samples collected during 2 months after application.

Pesticides movement into streams is well documented, but movement into ground water is not well researched. During movement from streamwater into ground water, concentrations should be reduced considerably for several reasons. Infiltrating pesticides must pass through several physical barriers or layers before reaching ground water. As they pass through each layer, they are degraded, diluted, and metabolized. Surface water provides a medium for dilution, hydrolysis, and photolysis. Aquatic vegetation can metabolize pesticides. Microbes associated with coarse and fine particulate organic matter found naturally in streams also metabolize pesticides.

In order for water on the soil surface to carry pesticides into ground water, it must pass through the soil column. Here again, processes work to reduce the potential for pesticides to reach ground water. Pesticides percolating through the soil column are adsorbed to soil particles, reducing the amount reaching the ground water. Pesticides adsorbed onto soil particles may be irreversibly bound, released slowly, or further metabolized by microbes. Once pesticides reach ground water, they may degrade further. Cavalier and others (1991) found that microbes degraded herbicides, including 2,4-D, in ground water.

Thus, ground water concentrations of pesticides should be considerably lower than observed in surface water. Funari and others (1995) reviewed the literature and reported the range of maximum ground water concentrations of pesticides, including those used in forestry, agriculture, home and garden, and on industrial rights-of-way. The maximum range of values for 2,4-D (0.0002 to 0.0495 mg per liter), hexazinone (0.009 mg per liter), and picloram (0.00063 to 0.049 mg per liter) are much lower than the HAL's for those compounds.

The National Water-Quality Assessment Program (NAWQA) conducted by the U.S. Geological Survey began in 1991. The focus of NAWQA is to identify nutrient and pesticide contamination of water throughout the United States. The 1999 NAWQA report (found at http://water.usgs.gov/pubs/circ/circ1225/index.html) makes little mention of forest sites or forestry pesticides, but concludes that: "Concentrations of nutrients and pesticides in streams and shallow groundwater generally increase with increasing amounts of agricultural and urban land in a watershed." The report focused on more than 50 major river basins and aquifers supplying water to more than 60 percent of the population and approximately half of the area of the United States. Few forestry pesticides other than 2,4-D were found in these basins or aquifers.

Even in predominantly agriculture areas, the report states:

> One of the most striking results for shallow groundwater in agricultural areas, compared with streams, is the low rate of detection for several high-use herbicides other than atrazine. This is probably because these herbicides break down faster in the natural environment compared to atrazine.

Atrazine is principally used in growing corn (maze). It has not been used on NFS land since 1992. While not directly addressing forestry pesticides and drinking water, these NAWQA conclusions support the above research findings and conclusions that ground water contamination by pesticides should be lower than observed for surface water. Because surface water contamination from forest sites treated according to label directions does not exceed HAL's, it is very unlikely that ground water contamination would exceed HAL's.

Several of the pesticides in table 13.2 have not been reported in water. They include chloropicrin, chlopyralid, dazomet, and thiram. Chloropicrin and dazomet are soil fumigants, which are gases in their active form and are used only for seedling production. Chlopyralid is a relatively new compound in the United States. Thiram is a dimethyl dithiocarbamate fungicide, principally used in forestry for seed protection.

There is very little water-quality data for pesticides used in nursery disease control and soil fumigation. More than 71 percent of fungicides and fumigants used on NFS land are applied in nurseries. Intense use in a nursery may result in localized ground water contamination. Three pesticides (chloropicrin, dazomet, and methyl bromide) make up this group of intensively used agents. Chloropicrin is toxic to plants and is used in combination with other chemicals for fumigating seedbeds. Dazomet, a soil fumigant, is a gas and is relatively insoluble in water (3 grams per liter). However, dazomet is unstable in water and quickly breaks down into methyl isothiocyanate (MITC), formaldehyde, monomethylamine, and hydrogen sulfide. All are toxic, but the most toxic is MITC. The RfD for formaldehyde is 0.2 mg per kilogram per day. However, EPA has classified formaldehyde as a compound of medium carcinogenic hazard to humans. Methyl bromide is very toxic. Data are insufficient to determine whether frequent use of these three pesticides adversely impacts water quality, either locally or over an expanded area.

Reliability and Limitation of Findings

Most data reviewed in this chapter come from scientific literature. The data listed in table 13.3 and derived from Larson and others (1997) were extracted from in-house reports from the U.S. Geological Survey, the EPA, State, and local governmental departments for the environment and scientific literature. Reports published in scientific literature are the most reliable because they were subject to peer review and scrutiny for validity of methods, completeness of data, and interpretation of the data. Monitoring data from in-house publications and reports may be less reliable.

Some variability in results of individual studies is due to regional soil and climate differences. In the South, infiltration rates on many forestry sites are generally low, owing to the highly eroded condition of the soils. Here, precipitation intensity frequently exceeds infiltration rates, producing overland flow on newly prepared sites. Overland flow may lead to much higher pesticide concentrations in stormflow than in other areas of the country with much higher infiltration rates. Very high infiltration rates are typical of soils in the Pacific Northwest. Therefore, if streams are protected by buffers, broadcast application of pesticides generally results in stream contamination either via direct application or through baseflow contributions. In general, levels of contamination are lowest where infiltration rates are highest.

Care must always be exercised in extrapolating data from local studies on drinking water to a regional or larger scale. However, three strategies of worst-case scenarios used in these studies mitigate against high levels of uncertainty: (1) several studies have investigated the impacts of pesticides applied directly to surface water; (2) several studies have investigated the impacts on water of pesticides applied at several times the prescribed rate; and (3) most of the studies conducted specifically on forestry sites treated the entire catchment from which water samples were taken, resulting in samples with levels of pesticide contamination greater than are likely to occur anywhere downstream.

Research on the impacts of pesticides applied directly to surface water used the worst-case scenario for forest operational treatments in which pesticide was applied at normal rates directly to surface water (ponds and streams). These studies did not find any contamination of water at levels above the HAL for any pesticide studied. Research on aquatic impacts from pesticides applied at several times the labeled rate used the worst-case scenario for operational treatments where an area might receive multiple applications in error or where small spills occurred. In these studies, HAL's were exceeded by only a few percent and only briefly, usually for less than a few hours. Both worst-case scenarios just described were combined with the third worst-case scenario in which all sampling was conducted on surface water found within the treated area. In this case, most of the water was in small pools or ephemeral to first-order streams. While ephemeral to first-order streams or pools are unlikely to be drinking water sources because of low yield, they do represent water most likely to be severely contaminated from normal forest pesticide applications. Even these waters were not contaminated at levels exceeding HAL's, except where pesticide was applied at several times the labeled rate as described above.

In addition, data on contamination of water for the pesticides in table 13.3 have been taken from a number of studies conducted in North America and the findings are generally similar. These studies have, with a few exceptions, confirmed the absence of significant contamination of drinking water. The exceptions were those cases in which a pesticide was applied directly to water, and the high concentrations observed in those studies were at or only slightly above drinking water standards. These high concentrations lasted only a few hours at most before dropping well below current HAL's. It is clear from the available literature that use of pesticides in strict accordance with label directions on NFS land cannot be expected to contribute significantly to ground water or drinking water contamination. It is also clear that pesticides, unless clearly labeled for aquatic uses, must not be applied directly to water, and that pesticides should be used around water resources, which are particularly sensitive only after careful consideration of the ramifications.

Limitations of the data are obvious for the few chemicals that have not been investigated. We need data on them as well as other chemicals about which little information is available. Additional limitations include lack of sufficient testing for health effects as indicated in table 13.2. The question of cumulative toxicological effects has not been addressed for any of the pesticide mixtures utilized in modern forest management.

Research Needs

Several issues related to vegetation management need additional research.

1. One issue is impacts of frequent, repeated use of fungicides and fumigants in nursery operations on nearby water quality.
2. Another issue is effectiveness of buffer width and composition. There is too little information on the processes and interactions of site-specific characteristics with pesticide chemistry that permit buffers to mediate against contamination of streams and surface waters in general. These processes and the interactions of pesticide chemistry with site-specific conditions must be identified and understood so managers can design and install optimally functional buffers to protect the water resource and its associated aquatic ecosystem.

3. A third issue is that several pesticides in table 13.3 lack published reports relating their use to occurrence in water.
4. Still another issue is the effects of commonly used pesticide mixtures, as opposed to single compounds, on the water resource.

Key Points

Relative to agricultural, urban, and other uses of pesticides, very small amounts are used on NFS land. Further, the use patterns for any specific piece of land are infrequent, except in the case of vegetation protection from pathogens, animals, and insects where annual treatment may be required, especially in greenhouses and nurseries. Pesticides used on NFS land are also used around the World to accomplish management goals similar to those on NFS land, but often in a much more intensive way. Even with the widespread use of pesticides in North America, those typically used on NFS land have not been identified in surface or ground water at sufficiently high concentrations as to cause drinking water problems. Their rapid break down by physical, chemical, and biological routes coupled with use patterns precludes the development of water contamination problems unless they are applied directly to water. Even though these same pesticides are used around homes, in urban and in agricultural settings, their use in forest management is still controversial in the public arena. Therefore, their use should be carefully planned and all agency, local, State, and Federal laws should be followed. It is especially important to follow all label directions because pesticide labels are legal documents specifying Federal laws pertaining to their use. Best management practices should be carefully adhered to and use around drinking water supplies should be avoided, except where permitted by the label. Wherever pesticides are used, precautions should always be taken to protect drinking water sources from contamination.

Literature Cited

Bouchard, D.C.; Lavy, T.L.; Lawson, E.R. 1985. Mobility and persistence of hexazinone in a forest watershed. Journal of Environmental Quality. 14: 229–233.

Bruce, B.W.; McMahon, P.B. 1996. Shallow ground-water quality beneath a major urban center. Journal of Hydrology. 186: 129–151.

Bush, P.B.; Michael, J.L.; Neary, D.G.; Miller, K.V. 1990. Effect of hexazinone on groundwater quality in the Coastal Plain. Proceedings of the Southern Weed Science Society; 1990 January 15–17; Atlanta. Champaign, IL: Southern Weed Science Society. 43: 184–194.

Bush, P.B.; Neary, D.G.; Taylor, J.W. 1988. Effect of triclopyr amine and ester formulations on groundwater and surface runoff water quality in the Coastal Plain. Proceedings of the Southern Weed Science Society; 1988 January 18–20; Tulsa, OK. Champaign, IL: Southern Weed Science Society. 41: 226–232.

Cavalier, T.C.; Lavy, T.L.; Mattice, J.D. 1989. Assessing Arkansas ground water for pesticides: methodology and findings. Ground Water Monitoring Review. 9: 159–166.

Cavalier, T.C.; Lavy, T.L.; Mattice, J.D. 1991. Persistence of selected pesticides in ground-water samples. Ground Water. 29: 225–231.

Comerford, N.B.; Neary, D.G.; Mansell, R.S. 1992. The effectiveness of buffer strips for ameliorating offsite transport of sediment, nutrients, and pesticides from silvicultural operations. Tech. Bull. 631. Research Triangle Park, NC: National Council of the Paper Industry for Air and Stream Improvement. 48 p.

Edwards, W.M.; Triplett, G.B., Jr.; Kramer, R.M. 1980. A watershed study of glyphosate transport in runoff. Journal of Environmental Quality. 9(4): 661–665.

Feng, J.C.; Thompson, D.G.; Reynolds, P.E. 1990. Fate of glyphosate in a Canadian forest watershed. 1: Aquatic residues and off-target deposit assessment. Journal of Agricultural and Food Chemistry. 38(4): 1110–1118.

Funari, E.; Donati, L.; Sandroni, D.; Vighi, M. 1995. Pesticide levels in groundwater: value and limitations of monitoring. In: Vighi, M.; Funari, E., eds. Pesticide risk in groundwater. Boca Raton, FL: Lewis Publishers. 275 p.

Holmes, S.B.; Millikin, R.L.; Kingsbury, P.D. 1981. Environmental effects of a split application of Sevin–2–oil. Rep. FPM–X–46. Sault Ste. Marie, Ontario: Canadian Forestry Service. 58 p.

Johnsen, T.N., Jr. 1980. Picloram in water and soil from a semiarid pinyon-juniper watershed. Journal of Environmental Quality. 9: 601–605.

Kolpin, D.W.; Kalkhoff, S.J.; Goolsby, D.A. [and others]. 1997. Occurrence of selected herbicides and herbicide degradation products in Iowa's ground water, 1995. Ground Water. 35: 679–688.

Koterba, M.T.; Banks, W.S.L.; Shedlock, R.J. 1993. Pesticides in shallow groundwater in the Delmarva Peninsula. Journal of Environmental Quality. 22: 500–518.

Larson, S.J.; Capel, P.D.; Majewski, M.S. 1997. Pesticides in surface water: distribution, trends, and governing factors. Ser. 5.3: Pesticides in the hydrologic system. Chelsea, MI: Ann Arbor Press. 373 p.

Leveille, P.; Legris, J.; Couture, G. 1993. Results of spot checks in lotic environments after spraying of glyphosate in forests from 1989 to 1991. Quebec, Canada: Ministere des Forets du Quebec. 21 p.

Michael, J.L.; Neary, D.G. 1993. Herbicide dissipation studies in southern forest ecosystems. Environmental Toxicology and Chemistry. 12: 405–410.

Michael, J.L.; Neary, D.G.; Fischer, J.B.; Gibbs, H.G. 1991. Metsulfuron in surface groundwater of a north Florida flatwoods. Proceedings of the Southern Weed Science Society; 1991 January 14–16; San Antonio, TX. Champaign, IL: Southern Weed Science Society. 44: 233–237.

Michael, J.L.; Neary, D.G.; Wells, M. 1989. Picloram movement in soil solution and streamflow from a Coastal Plain forest. Journal of Environmental Quality. 18: 89–95.

Michael, J.L.; Webber, E.C.; Bayne, D.R. [and others]. 1999. Hexazinone dissipation in forest ecosystems and impacts on aquatic communities. Canadian Journal of Forest Research. 29: 1170–1181.

Miller, J.H.; Bace, A.C. 1980. Streamwater contamination after aerial contamination of a pelletized herbicide. Res. Note SO–255. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 4 p.

Neary, D.G.; Bush, P.B.; Douglas, J.E.; Todd, R.L. 1985. Picloram movement in an Appalachian hardwood forest watershed. Journal of Environmental Quality. 14: 585–592.

Neary, D.G.; Bush, P.B.; Grant, M.A. 1986. Water quality of ephemeral streams after site preparation with the herbicide hexazinone. Forest Ecology and Management. 14: 23–40.

Newton, M.; Horner, L.M.; Cowell, J.E. [and others]. 1994. Dissipation of glyphosate and aminomethylphosphonic acid in North American forests. Journal of Agricultural and Food Chemistry. 42: 1795–1802.

Norris, L.A. 1975. Dicamba residues in streams after forest spraying. Bulletin of Environmental Contamination and Toxicology. 13: 1–8.

Norris, L.A. 1981. Phenoxy herbicides and TCDD in forests. Residue Reviews. 80: 65–135.

Pimentel, D.; Levitan, L. 1986. Pesticides: amounts applied and amounts reaching pests. BioScience. 36(2): 86–91.

Smith, N.J.; Martin, R.C.; St. Croix, R.G. 1996. Levels of the herbicide glyphosate in well water. Bulletin of Environmental Contamination and Toxicology. 57: 759–765.

Stanley, R.A.; Bates, A.L.; Bedsole, J.A.; Hall, T.F. 1974. Program diversity and control of Eurasian watermilfoil in Tennessee Valley reservoirs. In: Proceedings of the Weed Science Society of America; 1974 February 12–16; Las Vegas, NV: Champaign, IL: Southern Weed Science Society. 19 p.

Sundaram, K.M.S.; Szeto, S.Y. 1987. Distribution and persistance of carbaryl in some terrestrial and aquatic components of a forest environment. Journal of Environmental Science and Health, B (Pesticides, Food Contaminants and Agricultural Wastes). B22: 5, 579–599.

Thompson, D.G.; Staznik, B.; Fontaine, D.D. [and others]. 1991. Fate of triclopyr ester released in a boreal forest stream. Environmental Toxicology and Chemistry. 10(5): 619–632.

U.S. Department of Agriculture, Forest Service. 1998. Report of the Forest Service fiscal year 1997. Washington, DC: U.S. Department of Agriculture, Forest Service. 147 p.

U.S. EPA. 1988a. Picloram health advisory. Washington, DC: U.S. Environmental Protection Agency, Office of Drinking Water. 18 p.

U.S. EPA. 1988b. U.S. EPA integrated risk information system substance file–Ally. CASRN 74223–64–6. Washington, DC: U.S. Environmental Protection Agency. [Not paged]. [Available on the Web at: http://www.epa.gov/ngispgm3/iris/subst/0288.htm].

U.S. EPA. 1989. Drinking water health advisory. Pesticides. U.S. Environmental Protection Agency, Office of Drinking Water Health Advisories. Chelsea, MI: Lewis Publishers, Inc. 819 p.

U.S. EPA. 1990. Drinking water regulations and health advisories. Washington, DC: U.S. Environmental Protection Agency, Office of Drinking Water. 10 p.

U.S. EPA. 1992. U.S. EPA integrated risk information system substance file — Thiram. CASRN 137–26–8. Washington, DC: U.S. Environmental Protection Agency. [Not paged]. [Available on the Web at: http://www.epa.gov/ngispgm3/iris/subst/0267.htm].

U.S. EPA. 1993. Health advisories for drinking water contaminants. U.S. Environmental Protection Agency, Office of Water Health Advisories. Boca Raton, FL: Lewis Publishers. 261 p.

U.S. EPA. 1996. Hexazinone health advisory. Washington, DC: U.S. Environmental Protection Agency, Office of Drinking Water. 24 p.

U.S. EPA. 1997. Imazapyr (Arsenal) pesticide tolerances: final rule. Pesticide management education program. Ithaca, NY: Cornell University. [Not paged]. [Available on the Web at: http://pmep.cce.cornell.edu/profiles/herb-growthreg/fatty-alcohol-monuron/imazapyr/tol-imazapyr.html#top].

U.S. EPA. 1998a. Draft Resource Conservation and Recovery Act; waste minimization PBT chemical list; app. C: Waste minimization prioritization tool human noncancer scores. Washington, DC: U.S. Environmental Protection Agency, Office of Solid Waste, Waste Minimization Branch. [Not paged]. [Available on the Web at: http://www.epa.gov/epaoswer/hazwaste/minimize/chemlist/ap_c.pdf].

U.S. EPA. 1998b. Reregistration eligibility decision (R.E.D.) facts: Triclopyr. EPA–738–F–98–007. Washington, DC: U.S. Environmental Protection Agency, Office of Prevention, Pesticides and Toxic Substances (7508C). [Not paged]. [Available on the Web at: http://www.epa.gov:80/oppsrrd1/REDs/factsheets/2710fact.pdf].

Wood, J.A.; Anthony, D.H.J. 1997. Herbicide contamination of prairie springs at ultratrace levels of detection. Journal of Environmental Quality. 26: 1308–1318.

Part IV:

Effects of Grazing Animals, Birds, and Fish on Water Quality

Little green herons (Butorides virescens) are common to most water areas.
Photo by Bill Lea

Chapter 14

Domestic Grazing

John C. Buckhouse[1]

Introduction

Livestock grazing is a significant use of rangeland, and grazing practices can affect the quality of public drinking water sources. In general, activities that provide healthy range usually help maintain—or at least do not significantly degrade—water quantity and quality for domestic use. Rangeland management is not simply another phrase for livestock management. Rangeland often has climate, soils, topography, and precipitation characteristics or all that are unsuitable for intensive agriculture without irrigation or other intensive managerial inputs. Many rangelands have trees and are grazed.

Western rangelands are emphasized because most livestock grazing on natural forests and grasslands occurs in the West. Some also occurs in other regions. The publications cited here are meant to reflect physical and biological principles associated with water quality and livestock use. Therefore, while specific effects of livestock or water quality may vary, the underlying principles should be consistent.

Erosion and Sedimentation

Erosion and its consequence, sedimentation, are generally considered to be the number one problem associated with wildland watershed management.

Issues and Risks

Overgrazing weakens or kills vegetation, reducing soil cover and thereby accelerating surface erosion. With increased erosion, soil fertility declines and sediment yields increase. When significant amounts of sediment enter stream channels from rangeland, channels destabilize and widen, creating additional sources of sediment (U.S. Department of the Interior, Bureau of Land Management 1993, 1998). Sediment reduces water clarity and the oxygen-carrying capacity of the stream. Nutrients attached to sediment heighten the possibility of eutrophication. See chapter 2 for more discussion of sediment impacts on drinking water quality.

Findings from Studies

Considerable research is available on the relationships between livestock grazing and erosion and sedimentation (fig. 14.1). Several textbooks summarize the effects of livestock numbers, livestock types, timing of grazing, and animal distribution on vegetation and erosion (Holecheck and others 1989, Stoddart and others 1975). At high densities, grassland vegetation promotes production of soil organic material and increases infiltration rates (Buckhouse and Gaither 1982, Buckhouse and Mattison 1980). Therefore, grazing should be managed to maintain the density of vegetation, reducing erosion, and sediment yields.

In the riparian zone, the relationship of livestock grazing to streambank erosion has been studied (Bohn and Buckhouse 1985, Buckhouse and Bunch 1985, Buckhouse and others 1981). These researchers found it is possible to manage livestock grazing in ways that enhance riparian vegetation and protect streambanks. Grazing throughout the growing season harms vegetation and results in increased streambank sloughing. Conversely, grazing that is timed to accommodate plant growth and physiology can have positive effects on streams and the quality of water in them. In Oregon, for example, early-season grazing enhanced riparian shrub growth (Buckhouse and Elmore 1997).

J.M. Skovlin (1984) prepared an excellent review of livestock grazing research. He described effects on vegetation, streambank stability, and various aspects of water quality. He also compiled a large reference list and provided recommendations and prescriptions for grazing management. This document is an exhaustive look at many aspects of grazing management.

Reliability and Limitations of Findings

This work has been repeated and verified in several locations by several researchers. It is known that loss of plant

[1] Professor, Oregon State University, Department of Rangeland Resources, Corvallis, OR.

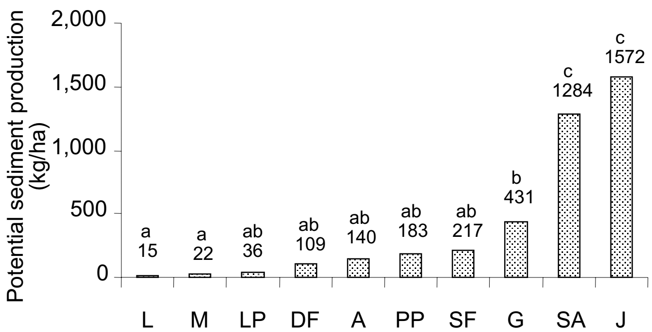


Figure 14.1—Potential sediment production in 10 Blue Mountain ecosystems in Oregon. Ecosystems: L = larch, M = meadow, LP = lodgepole pine, DF = douglas-fir, A = alpine, PP = ponderosa pine, SF = spruce-fir, G = grassland, SA = sagebrush, J = juniper (Buckhouse and Gaither 1982). Different lower-case letters indicate differences in statistical significance (P<0.10).

cover increases sediment yield. The key to applying findings is to think in terms of plant physiology and plant response to grazing. The goal of proper grazing is to maintain vegetation and soil organic material, preserving the soil's ability to infiltrate water, resist erosion, and store and slowly release precipitation.

Managerial success depends on understanding ecosystems, the vegetation present or desired, and management objectives, and then prescribing herbivory appropriate to the site. The Forest Service and the U.S. Department of the Interior, Bureau of Land Management, have cooperated to develop Proper Functioning Condition (PFC), a first-cut methodology to monitor wildland streams. Wayne Elmore and his riparian team have based their approaches on these concepts, which have been widely adopted (U.S. Department of the Interior, Bureau of Land Management 1998). However, there have been no systematic studies of how effective these methods are for protecting the quality of public drinking water sources.

Research Need

Refinement of the relationships among herbivory, vegetation, and sediment production on specific sites in specific ecosystems needs more research.

Bacteria and Protozoa

Waterborne, pathogenic bacteria and protozoa have long been recognized as causes of human disease. Many diseases can be transmitted only among members of the same species, but a significant number can be transmitted to hosts of different species. The concern is that contamination of the drinking water supplies by humans and by a variety of animal species poses hazards to human health.

Issues and Risks–Fecal Bacteria

Escherichia coli is a ubiquitous bacterium found in the gut of all warm-blooded animals. It is used as an indicator species in bacteriological testing. Other pathogenic organisms are difficult to trap, difficult to analyze, and expensive to process. As a consequence, testing for fecal coliform bacteria has become the accepted surrogate sampling protocol, with the understanding that if fecal coliforms are present, then pathogens are potentially present. *Escherichia coli* was long considered to be benign. However, in recent years several pathogenic strains of *E. coli* have developed and gained national attention due to the health risk they pose.

Findings from Studies (Bacteria)

Walter and Bottman (1967) reported on the two tributaries of the watershed supplying the city of Bozeman, MT, with its drinking water. One tributary was fenced and human activity was limited. A corresponding tributary was not restricted from human entry. The fenced watershed consistently had higher fecal bacteria loads than did the open watershed. After considerable monitoring and study, it was realized that the closed watershed had become a de facto refuge for wildlife. The increased animal use resulted in the higher fecal bacteria numbers in streamwater.

Coltharp and Darling (1973) studied three neighboring watersheds with different combinations of animals grazing and browsing: wildlife, sheep and wildlife, and cattle and wildlife. They found the lowest numbers of bacteria in the streams in the wildlife-only watershed. The sheep-and-wildlife watershed was next lowest, and the cattle-and-wildlife watershed had the highest bacteria counts. There was high statistical variation in the numbers, but in all three strategies, the bacterial counts were <100 per 100 milliliters (ml). The authors attributed the differences to animal numbers and distribution. The cattle tended to congregate near the stream, while the sheep were herded and spent more time in the uplands.

Bohn and Buckhouse (1981, 1983) sampled water in northeastern Oregon and found similar results. On watersheds with a resident deer and elk population, where grazing by livestock was excluded for several months during the winter, the bacteria numbers were low (10 to 20 per 100 ml). At the end of the summer, after livestock had grazed the area for several months, the bacteria numbers were slightly elevated (20 to 40 per 100 ml).

Research in central Oregon studied the fate of the coliform bacteria (Biskie and others 1988; Larsen and others 1988, 1994; Moore and others 1988; Sherer and others 1992). In this Great Basin rangeland, they discovered that statistically only 5 days per year experienced enough precipitation to produce overland flows. Consequently, the probability of washing fecal material into streams was relatively low. Cows defecated approximately 11 times a day, but less than one of these defecations landed in the stream directly or within 1 meter of the stream where it might be washed into the stream. Further experiments were conducted to determine the fate of bacteria defecated directly into the stream. Ninety percent of these bacteria precipitated to the stream bottom and attached to sediments. Sediment samples collected over the next several weeks showed that 90 percent of the lodged bacteria had died within 40 days (Biske and others 1988; Larsen and others 1988, 1994; Moore and others 1988; Sherer and others 1992).

The same team of researchers studied the effect of a strategically placed watering trough on livestock use of a flowing stream in both summer and winter (Clawson and others 1994, Miner and others 1992). When snow covered the ground, cattle were fed hay, and 95 percent of them used the ground water-fed trough, as opposed to 5 percent that used the stream. It is speculated that the warmer ground water (approximately 50 °F or 10 °C) held much greater appeal to the cattle than did the 32 °F (0 °C) creek. In the summer, riparian zones provided lush vegetation, shade, and water. The trough relieved some impact during the summer, but not as dramatically as during the winter. In late summer, when most of the available vegetation near the riparian zone had been consumed, approximately 25 percent of the livestock drank from the trough, while the remaining 75 percent preferred the stream.

Findings from Studies (Protozoa)

Giardia and *Cryptosporidium* have drawn considerable attention recently. Both are debilitating to humans and can be carried by a wide variety of warm-blooded animals from waterfowl to rodents; from deer, elk, and beaver to livestock (see appendix B and chapter 15). Both *Giardia* and *Cryptosporidium* have been known for decades, but only recently have routine testing following gastrointestinal complaints from citizens been conducted. It is probable that what is now diagnosed as *Cryptosporidium* would at an early time have been seen simply as flu or food poisoning.

Cryptosporidium oocysts have been found in association with both wild and domestic animals. Calves consistently shed greater numbers of oocysts than do older animals (Atwill 1996). Apparently by 4 months of age, calves develop a resistance and the number of oocysts shed is dramatically reduced. An alternative to livestock exclusion from areas where *Cryptosporidium* may be a concern is to ensure that livestock grazing the watershed are older than 4 months. The relationship between oocysts and age of wild animals is unknown and represents a research question.

Reliability and Limitations of Findings

The fecal bacteria research is reliable and has been consistent over time. The *Cryptosporidium* work, while compelling, is relatively recent.[2] Tate and his colleagues at the University of California, Davis, continue to investigate these relationships, the biology, and the management of *Cryptosporidium*. Conceptually, these experiments were solid. Site-specific relationships dictate specific biological responses. However, regional climatic and temperature differences may influence the range of variability inherent in these relationships and to date little is known about wild animals except that they carry the organisms and shed them in their feces. Water from wildlands including wilderness, can contain pathogens that cause human disease if the water is not adequately treated for purification.

Research Needs

Cryptosporidium research is still in its infancy. Further studies dealing with the origin and fate of the organisms are needed. Very little is known about pathogenic *E. coli* in livestock under range conditions, and further investigation is needed.

Chemical and Nutrient Impacts

A number of chemical compounds are associated with sediment transport. Rangelands are sometimes treated with herbicide or fertilizer compounds, but the primary chemical or nutrient problems are with phosphate (PO_4^{+3}) and nitrate (NO_3^{-1}) associated with eroding soils and fecal material.

Issues and Risks

Cycling of phosphate and nitrate are essential to plant growth. When vegetation is grazed, potential problems exist where phosphates and nitrates from feces or attached to

[2] Personal communication. 1999. Kenneth Tate, Professor, Department Agronomy and Range Science, University of California, Davis, CA 95616.

erosion particles that find their way to a stream. Excessive levels of these nutrients in streams can produce algal blooms and eutrophication. See chapter 2 for discussion of nutrient impacts on drinking water.

Findings from Studies

Frequently phosphate and nitrate reach streams in association with sediments. See chapter 3, which discusses overland flow, erosion, and sediment transfer.

When fecal material enters a stream, phosphate and nitrate concentrations rise, but wetlands can reduce these concentrations. On the Wood River system in Oregon, a nutrient loading was studied on streams originating from the Crater Lake National Park, traversing national forest, then private grazing land, and emptying into a lake listed as hypo-eutrophic. The concern had been that nutrient loading would be exacerbated when the water flowed through the grazed land due to fecal contamination. The data did not bear out this fear. In fact, the phosphate and nitrate levels decreased.[3] It was speculated that the wetlands in the system represented a natural nutrient sink. In wetlands, chemical processes associated with anaerobic conditions reduced phosphate and nitrate concentrations. Furthermore, wetland plants take up nutrients from the aqueous system. If animals eat wetland vegetation, nutrients are consumed. Thus, these nutrients are not redistributed back into the water as when the plant senesces in place and dies. Apparently the livestock grazing the wetland consumed the vegetation and its nutrients, and later redistributed the nutrients away from the stream in their feces. The result was the observed decline in water-borne nutrient concentrations.

Reliability and Limitations of Findings

The relationship between nutrient loading and sediment is clear. The relationship between livestock grazing of wetlands and the possibility of nutrient loading abatement is not proven. The relationships among erosion, sedimentation, and nutrient loading are universal and are bases for erosion control worldwide. The possibility of using livestock for biological control of weeds, for improving plant communities, for promoting species that encourage infiltration and reduce overland flow, and even for reducing nutrient loading (Bedell and Borman 1997) are based on solid research and management experience. These concepts are predicated on the ability of vegetation and organic material to enhance infiltration, which reduces overland flows and subsequent erosion. The concept of using livestock to harvest nutrient-rich wetland vegetation is logical but not tested.

Research Needs

Further research is needed on the relationships between livestock harvesting of streamside vegetation and nutrient loading of streams.

Key Points

In rangeland, grazing can affect drinking water. Water quality can be protected by nurturing upland and riparian vegetation, which can increase the soil's infiltration capacity and reduce surface runoff. By understanding the relationships between plant physiology and animal herbivory, one can tailor grazing practices to enhance infiltration. Both timing and intensity of grazing must be managed. Increased infiltration reduces sediment yield, bacterial counts, nutrient concentrations, and water temperatures. Published data clearly indicate that improper, abusive grazing can degrade the quality of public drinking water sources. It is also clear that proper, prescriptive grazing can produce positive environmental benefits, and research is needed to develop and test methods to accomplish them.

Literature Cited

Atwill, E.R. 1996. Assessing the link between rangeland cattle and waterborne *Cryptosporidium parvum* infection in humans. Rangelands. 18: 48–51.

Bedell, T.E.; Borman, M.M. 1997. Watershed management guide for the interior Northwest. Ext. Man. 8436. Corvallis, OR: Oregon State University Extension Service. 84 p.

Biskie, H.A.; Sherer, B.M.; Moore, J.A. [and others]. 1988. Fate of organisms from manure point loading into a rangeland stream. Pap. 88–2081. American Society of Agricultural Engineers: 1–14.

Bohn, C.C.; Buckhouse, J.C. 1981. Interpreting coliform counts in northeast Oregon rangeland streams. Spec. Prog. Rep. 620. Research in rangeland management. Corvallis, OR: Oregon State University, Agricultural Experiment Station: 15–17.

Bohn, C.C.; Buckhouse, J.C. 1985. Coliforms as an indicator of water quality in wildland streams. Journal of Soil and Water Conservation. 40: 95–97.

Buckhouse, J.C.; Bohn, C.C. 1983. Response of coliform bacteria concentration to grazing management. Spec. Prog. Rep. 682. Corvallis, OR: Oregon State University, Agricultural Experiment Station: 1–8.

[3] Personal communication. 1997. Ronald Hathaway and Rodney Todd, Extension Agents, Klamath County Extension Office, Klamath Falls, OR 97601.

Buckhouse, J.C.; Bunch, T.E. 1985. Riparian erosion inside and outside of exclosures on Mill and McKay Creeks: a validation of management. Spec. Prog. Rep. 743. Research in range management. Corvallis, OR: Oregon State University, Agricultural Experiment Station: 29–30.

Buckhouse, J.C.; Elmore, W. 1997. Grazing practice relationships: predicting riparian vegetation response from stream systems. In: Bedell, T.E.; Borman, M.M., eds. Watershed management guide for the interior Northwest. Ext. Man. 8436. Corvallis, OR: Oregon State University, Extension Service: 47–52.

Buckhouse, J.C.; Gaither, R.E. 1982. Potential sediment production within various vegetative communities found in the Blue Mountains of Oregon. Journal of Soil and Water Conservation. 37: 120–122.

Buckhouse, J.C.; Knight, R.W.; Skovlin, J.M. 1981. Streambank erosion and ungulate grazing relationships. Journal of Range Management. 34: 339–340.

Buckhouse, J.C.; Mattison, J.L. 1980. Potential soil erosion of selected habitat types of the high desert region of central Oregon. Journal of Range Management. 33: 282–286.

Clawson, J.; Buckhouse, J.C.; Miner, J.R.; Moore, J.A. 1994. An evaluation of off-site water development and fence configurations designed to ameliorate water quality impacts by livestock [Abstract]. Corvallis, OR: Oregon State University, Oregon Academy of Science. 1 p.

Coltharp, G.B.; Darling, L.A. 1973. Livestock grazing—a nonpoint source of water pollution in rural areas? In: Proceedings of the symposium on rural environmental engineering; 1973 September 15; Minneapolis. Burlington, VT: University of Vermont. 15 p.

Holechek, J.L.; Piefer, R.D.; Herbel, C.H. 1989. Range management principles and practices. Englewood Cliffs, NJ: Prentice-Hall. 501 p.

Larsen, R.E.; Buckhouse, J.C.; Moore, J.A.; Miner, J.R. 1988. Rangeland cattle and manure placement: a link to water quality. Proceedings of the Oregon academy of science; 1988 February 28; Corvallis, OR. Corvallis, OR: Oregon State University. 24: 7–15.

Larsen, R.E., Miner, J.R.; Buckhouse, J.C.; Moore, J.A. 1994. Water quality benefits of having cattle manure deposited away from streams. Bioresource Technology. 48: 113–118.

Miner, J.R.; Buckhouse, J.C.; Moore, J.A. 1992. Will a water trough reduce the amount of time hay-fed livestock spend in the stream (and therefore improve water quality)? Rangelands. 14: 35–38.

Moore, J.A.; Buckhouse, J.C.; Miner, J.R. 1988. Water quality impacts of alternate rangeland management systems. Comp. Rep. 85–CSRS–2–2718. [Location of publisher unknown]: U.S. Department of Agriculture, Cooperative State Research Service. 125 p.

Odum, E.P. 1971. Fundamentals of ecology. Philadelphia: W.B. Saunders Co. 574 p.

Sherer, B.M.; Miner, J.R.; Moore, J.A.; Buckhouse, J.C. 1988. Resuspending organisms from a rangeland stream bottom. Transactions of the American Society of Agricultural Engineers. 31: 1217–1222.

Sherer, B.M.; Miner, J.R.; Moore, J.A.; Buckhouse, J.C. 1992. Indicator bacterial survival in stream sediments. Journal of Environmental Quality. 21: 591–595.

Skovlin, J.M. 1984. Impacts on wetlands and riparian habitat: a review of our knowledge. In: Developing strategies for rangeland management. A report prepared by the Committee on Developing Strategies for Rangeland Management, National Research Council, National Academy of Sciences. Boulder, CO: Westview Press: 1001–1104.

Stoddart, L.A.; Smith, A.D.; Box, T.W. 1975. Range management. New York: McGraw-Hill, Inc. 532 p.

Stringham, T.K.; Buckhouse, J.C.; Krueger, W.C. 1998. Stream temperatures as related to subsurface waterflows originating from irrigation. Journal of Range Management. 51: 88–90.

U.S. Department of the Interior, Bureau of Land Management. 1993. Riparian area management: process for assessing proper functioning condition. Tech. Ref. 1737–9. Denver: Bureau of Land Management Service Center. 51 p.

U.S. Department of the Interior, Bureau of Land Management. 1998. Riparian area management: a user guide to assessing proper functioning condition and supporting science for lotic areas. Tech. Ref. 1737–15. Denver: National Applied Resource Sciences Center. 127 p.

Walter, W.G.; Bottman, R.P. 1967. Microbiological and chemical studies of an open and a closed watershed. Journal of Environmental Health. 30: 157–163.

Chapter 15

Wildlife

Arthur R. Tiedemann[1]

Introduction

Numbers of large herbivores—mainly deer and elk—are increasing throughout the United States (Riggs and others, in press) with a commensurate increase in potential for them to have an effect on the quality of drinking water sources. In general, activities that provide healthy habitat usually help maintain—or at least do not significantly degrade—water quantity and quality for domestic use. However, contamination of surface waters by wild herbivores and disease-organism transmission among domestic herbivores and wide-ranging large herbivores are major issues that must be addressed by land managers. Emphasis of this chapter is the role of large, wild herbivores, such as deer, elk, and moose on microbiological quality of wildland surface water. Information is also provided on the role of beavers and muskrats.

Issues and Risks

Presence of disease-causing organisms in wildland surface waters is the most critical aspect of problems related to the influence of wildlife on water quality. There is no doubt that the human pathogens *Giardia* spp. and *Cryptosporidium* spp. (appendix D), and pathogenic *Escherichia coli* (*E. coli*) are carried by and can be spread by a wide variety of wildlife.

The impact of large, wild herbivores, such as deer, elk, and moose, on water quality is an elusive and difficult problem. These animals range widely and unpredictably, and their densities and movements relative to surface waters are very difficult to quantify.

Presence and numbers of fecal coliform (FC) in a given volume of streamwater [usually number per 100 milliliters (ml)] are used as indicators of the potential for presence of disease organisms. At FC counts between 1 and 200 per 100 ml of water, the percentage occurrence of salmonella disease organisms is 28 percent (Geldreich 1970). Occurrence of salmonella increases to 85 percent for FC counts of 200 to 2,000 per 100 ml of streamwater and to 98 percent at FC levels in excess of 2,000 per 100 ml. Bohn and Buckhouse (1985) comprehensively reviewed the use of FC as an indicator of wildland water quality. Use of FC counts as an indicator of disease bacteria has several drawbacks. For example, it is not a satisfactory indicator for *Giardia.*

Fecal streptococcus (FS) counts are also used as indicators of the presence of disease organisms in water (Sinton and others 1993), but there are no established standards as there are with FC.

The ratio of FC to FS is one measure of the source of bacterial contamination of wildland surface water (Geldreich 1967). Geldreich (1976), Geldreich and Kenner (1969), and Van Donsel and Geldreich (1971) established ranges of FC to FS—in feces for humans, >4; cattle, 1.2 to 0.8; cattle and wildlife mixed, 0.08 to 0.04; and wildlife, <0.04. Tiedemann and Higgins (1989) and Tiedemann and others (1988) applied the concept to wildland watersheds where several watersheds received only wildlife use and others received varying degrees of cattle use and wildlife use. For watersheds with no cattle use, the FC to FS ratio was <0.04 for 90 percent of the samples collected. On watersheds with intensive cattle use of 7 acres [2.8 hectares (ha)] per animal unit month (aum), the FC to FS ratio in 75 percent of the samples was between 1.2 and 0.8. On watersheds with lower intensities of cattle use [19 to 20 acres (7.7 to 8.1 ha) per aum], the ratio suggested a mixture of cattle and wildlife use. Thus, the ratio established by Geldreich appears to have potential as a tool to separate wildlife and cattle sources of pollution. Baxter-Potter and Gilliland (1988) also concluded that the ratio was useful for distinguishing contaminant sources if the pollution had not aged in the stream.

Recent efforts have focused on determination of the presence and sources of actual disease organisms in surface water. Outbreaks of severe, bloody diarrhea associated with *E. coli* [O antigen 157 and H antigen 7 (O157:H7)] (Armstrong and others 1996) and numerous reports of *Giardia* infection (Moore and others 1993) have been

[1] Scientist Emeritus, USDA Forest Service, Pacific Northwest Research Station, Wenatchee, WA.

responsible for emphasis on these organisms. Most cases of *E. coli* O157:H7 infection and death have been caused by consumption of infected meat (Armstrong and others 1996), including deer jerky (Keene and others 1997). However, multiple illnesses and deaths have occurred as a result of drinking contaminated municipal water (Swerdlow and others 1992). In the case of *E. coli*, until 1982, this organism had not been considered a disease-causing entity (Riley and others 1983, Wells and others 1983). As discussed previously, *E. coli* was formerly considered to be benign and an indicator organism for the presence of organisms that actually cause diseases.

Severe illness and deaths resulting from *E. coli* O157:H7 have prompted accelerated research to determine presence and quantities occurring in various animal species including wildlife. This is usually done by culture of samples of fresh fecal material (Atwill and others 1997, Goodrich and others 1973, Monzingo and Hibler 1987), entrail samples from harvested animals (Atwill and others 1997, Frost and others 1980), or samples from live animals (Erlandsen and others 1990). Transmission among domestic and wildlife species (especially those with wide ranges) is a crucial facet of this problem with far-reaching implications for drinking water quality. Large herbivores, such as deer, elk, and moose, are without boundaries and may serve as a vector for transmission of disease organisms among domestic livestock in pastures or on rangelands. This concern is reinforced by results of Sargeant and others (1999), who found similar genetic strains of *E. coli* O157:H7 in cattle and free-ranging white-tailed deer sharing the same pasture. Both types of animals serve as reservoirs for *E. coli* O157:H7. Although their conclusions were directed at consumption of deer meat, their results suggest that we should be alert to potential problems with the quality of surface drinking water sources.

Free-ranging white-tailed deer have also been shown to be potential sources of contamination of *Giardia* and *C. parvum* (Rickard and others 1999). These authors concluded “the abundance of water throughout the State (Mississippi) coupled with an overpopulation of white-tailed deer indicates this cervid may pose a threat to surface water of this area.”

Relations among wild herbivores and water quality are highly complex involving the physical and vegetal nature of the landscape, characteristics of surface water, the species of wildlife, interrelations among wildlife and domestic animals, and the disease or indicator organism involved. Land management activities that tend to concentrate wildlife or result in increased densities also will influence water quality. Landscape characteristics that would tend to influence the effect of wild herbivores on water quality would be those that encourage or discourage high animal concentrations near surface water, such as extremely steep terrain or very dense vegetation. Stream characteristics that influence organism concentrations are discharge, turbidity, conductivity, pH, and temperature. Diurnal fluctuations of bacterial counts are related to discharge, but evidence is contradictory. Bohn and Buckhouse (1985) provide detailed analyses of watershed characteristics that influence FC numbers in streamwater. According to these authors, bacterial counts generally increase with increasing discharge. Increased counts are apparently the result of increased flushing of streambanks and associated fecal deposits. Sediment transport and associated organisms may also account for some of the increase in counts with increasing discharge. Counts of FC have been shown to be much higher in sediments than in surface water (Skinner and others 1984, Stephenson and Rychert 1982). Counts of FC seem to be inversely related to stream temperature (Geldreich and Kenner 1969). Extremes of pH also influence FC viability with 5.5 to 7.5 as an optimal range (McFeters and Stuart 1972). Type of wildlife species present is a major determinant of the type and number of disease organisms present in surface water. Wildlife closely associated with water, such as beavers and muskrats, may cause a protracted, relatively stable contamination problem that may be fairly predictable. Contamination by wider ranging wildlife, such as deer and elk, will likely be transient, sporadic, and somewhat unpredictable. Type of organism present also depends to some degree on the species of wildlife present. *Giardia*, for example, appears to be more closely tied to presence of water-associated wildlife, such as beavers and muskrats, than deer and elk (Monzingo and Hibler 1987). This does not exclude deer and elk as an element of the cycle of the organism or as a transmission mechanism. The disease-causing organism *E. coli* O157:H7 appears to be more closely related to ruminants than to animals with simple stomachs (Armstrong and others 1996).

Persistence of fecal organisms in surface water is an important consideration. In the case of domestic animals, FC levels in streamwater remain high for many months after cattle are removed (Jawson and others 1982, Tiedemann and others 1987). We don’t know if the same is true for wildlife effects on water quality. Fecal coliform do survive for long periods—at least 1 year in feces of both domestic (Clemm 1977) and wild herbivores (Goodrich and others 1973).

Complexity also arises from the rigorous sample collection, transport, and analytical procedures that must be followed to obtain accurate results. Isolation of actual disease organisms is a complicated laboratory procedure that may also be very costly. Sampling schedules must be carefully adhered to.

Land treatment measures such as forest thinning, prescribed burning, shrub control, seeding, and fertilization have the potential to alter wild herbivore-use patterns. Some treatments, such as prescribed burning, are done to improve wildlife habitat. These treatments have direct effects on water quality that are discussed in chapters 10, 12, and 14 of this report. Because the treatments have the potential to also alter wild herbivore-use patterns, there is a potential for indirect effects on water quality. If these habitat modifications concentrate animals near streams and near domestic water supply withdrawals, water-quality impacts could be serious. However, information is lacking on effects of wild herbivores on water quality associated with changes in habitat and resultant alterations in herbivore-use patterns.

Findings from Studies

Very little research has been conducted on large, wild herbivores or beavers and muskrats as the identifiable sources of fecal contamination of wildland surface water. Results of studies of maximum levels of FC and *Giardia* resulting from wild herbivore use are portrayed in table 15.1. Walter and Bottman (1967) provide some of the earliest information on effects of wild herbivores on water quality. They studied FC counts in two watersheds that serve as a source of municipal water for Bozeman, MT. One watershed was used by recreationists; the other was fenced and patrolled to exclude the public and livestock. Thus, the only potential source of FC in the closed watershed was wildlife. Deer, elk, and moose were present in the closed watershed in undetermined numbers. They found maximum FC counts exceeding 200 per 100 ml of streamwater in the closed watershed. Fecal coliform counts were actually much higher in streamwater in the closed than in the open watershed, suggesting higher wildlife populations in the closed watershed—perhaps as a result of high levels of human use of the open watershed or watershed characteristics.

In another early study, Kunkle and Meiman (1967) measured maximum FC counts of 25 per 100 ml in streamwater from an area essentially free from human impact—very limited hiking or camping, and no domestic livestock grazing. Wildlife species were not listed, but it is safe to assume that the area is inhabited by mule deer.

Maximum FC (100+ per milliliter) and *Giardia* (1 per 100 ml) levels were also high in streamflow from a pristine, forested watershed in western Washington. These levels were presumably from wildlife contamination because human use was very light [5 to 30 days per year per 0.6 miles (1 kilometer) of stream]. Deer, elk, beavers, mountain beavers, river otters, and marmots were listed as possible sources of contamination.

Streamwater from pristine land in southern Finland was shown to contain high levels of FC (maximum, 268 per 100 ml of streamwater). The high counts apparently resulted from wild deer and moose use of the watershed (Niemi and Niemi 1991).

In eastern Oregon, forested watersheds that supported no domestic grazing had maximum FC counts in excess of 500 per 100 ml of streamwater (Tiedemann and others 1987). Deer and elk were the predominant large herbivores inhabiting the watersheds.

Maximum *Giardia* count observed by Monzingo and Hibler (1987) was 0.06 per 100 ml of streamwater in beaver ponds in Colorado.

Streamflow from watersheds without domestic livestock in Utah (Doty and Hookano 1974) and Wyoming (Skinner and others 1974) contained relatively high counts of FC—22 to 183 per 100 ml. Although FC origin was attributed to wildlife in both studies, there was no mention of which wildlife species were responsible. Recreational use in the two studies was limited to hiking.

Reliability and Limitation of Findings

Research to date lacks replication or detailed examination of species responsible, numbers of each species, and their distribution. Its results, however, indicate that wild herbivores pose a risk to drinking water quality.

Studies spanned a relatively broad geographical scope and suggest that drinking water quality may be a problem wherever there is contact between wild herbivores and surface water.

Changes in habitat, whether deliberate or uncontrolled, that attract wildlife to surface drinking water sources may increase the risk of introducing contaminants. Management that discourages animals from concentrating near streams that are sources of public drinking water may help alleviate potential contamination problems.

Table 15.1—Streamwater fecal coliform and *Giardia* responses to herbivore use

Location	Setting	Herbivore	Maximum counts observed[a]		Reference
			Fecal coliform	*Giardia* spp.[b]	
Colorado	Beaver ponds	Beaver	Not measured	0.021	Monzingo and Hibler 1987
Washington	Pristine forested watershed, low-level human use	Beaver, mountain beaver, otter, marmot	100+	1	Ongerth and others 1995
Southern Finland	Pristine lands	Elk, deer	268		Niemi and Niemi 1991
Eastern Oregon	Forested watersheds, no domestic livestock use	Deer, elk	500+		Tiedemann and others 1987
Montana	Municipal watershed, closed to public access	Deer, elk	200+		Walter and Bottman 1967
Colorado	Area essentially free of human impact, no domestic livestock use	Not indicated	25		Kunkle and Meiman 1967
Utah	Watersheds with no livestock use	Not indicated	36–183		Doty and Hookano 1974
Wyoming	Watershed open to hikers, no livestock use	Not indicated	22		Skinner and others 1974

[a] Counts per 100 milliliters of streamwater.
[b] Locations with no data in this column indicate that they were not measured.

Research Needs

1. Carefully controlled experiments are needed to determine the magnitude of the effect of wild herbivores on microbiological water quality. Such research must separate effects of domestic herbivores and humans from those that result from wildlife. Research must also document the species of herbivore, the amount of time that each spends near surface water, and organisms associated with fecal deposits. Carefully designed water sampling procedures must accompany these studies. Samples above, within, and below areas of activity of sufficient frequency to represent all hydrograph stages will be essential.
2. Landscape-scale satellite animal tracking experiments with large, wild herbivores, such as the Starkey Experimental Forest and Range studies in northeast Oregon (U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, La Grande, OR), provide opportunities to relate large herbivore movements by species to changes in water quality. Rationale and methodology of those studies could be used as a model for design of water-quality studies. Starkey's studies (Rowland and others 1997) could also serve as a model for research to determine relations among forest and rangeland treatments, wild herbivore-use patterns, and resultant changes in water quality.

3. Perhaps it would be appropriate to explore advanced technology to discourage large, wild herbivores, such as the technique described by Tiedemann and others (1999) for cattle. This technique would probably be appropriate primarily where wild herbivore contamination of surface water creates severe risks from drinking the water.

4. The literature shows that microbiological and epidemiological research on *Giardia* and *E. coli* O157:H7 is well underway. Hancock and others (1998), however, pose some problems that are urgently in need of resolution; *E. coli* O157:H7 may: (1) have multiple species in which it resides for long periods; (2) be capable of transiently colonizing many species; and (3) have an environmental source, such as sediments where it may be able to multiply prolifically. Similar questions should be raised with regard to *Giardia*.

Key Points

1. Presence of wild herbivores has the potential to influence microbiological quality of wildland surface water and to render it unsafe for drinking unless it is adequately purified. It is important for wildland managers to understand the potential for a problem to exist even though they cannot document the actual source of contamination.

2. Relationships between wild herbivores and water quality are highly complex involving physical and vegetal landscape characteristics, hydrologic parameters (discharge, pH, temperature), type of contaminating organisms, species and numbers of animals, use patterns of animals, mixing of domestic and wild herbivores, length of time since animals were present, and land treatment measures that have been implemented.

3. Potential for presence of disease organisms such as pathogenic *E. coli* O157:H7 and *Giardia* emphasizes the urgency to understand the relations between wild herbivores and water quality. No studies have related water quality to wild herbivore species, numbers, and land-use patterns.

4. Land treatment measures such as thinning, prescribed burning, seeding, and fertilization, whether designed specifically for wildlife habitat alteration or other management purposes may alter herbivore-use patterns and exert secondary effects on water quality. These effects are presently undocumented and poorly understood.

5. Free-ranging large, wild herbivores may be vectors for transmission of disease organisms among domestic livestock. Associated contamination of surface water and potential public health hazards create an urgency to understand these relations and effects on water quality.

Literature Cited

Armstrong, Gregory L.; Hollingsworth, Jill; Morris, J. Glenn, Jr. 1996. Emerging foodborne pathogens: *Escherichia coli* O157:H7 as a model of entry of a new pathogen into the food supply of the developed world. Epidemiological Reviews. 18: 29–51.

Atwill, Edward R.; Sweitzer, Richard A.; Das Gracas, C. [and others]. 1997. Prevalence of and associated risk factors for shedding *Cryptosporidium parvum* oocysts and *Giardia* cysts within feral pig populations in California. Applied and Environmental Microbiology. 63: 3946–3949.

Baxter-Potter, Wanda R.; Gilliland, Martha W. 1988. Bacterial pollution in runoff from agricultural lands. Journal of Environmental Quality. 17: 27–34.

Bohn, Carolyn C.; Buckhouse, John C. 1985. Coliforms as an indicator of water quality in wildland streams. Journal of Soil and Water Conservation. January-February: 95–97.

Clemm, David. 1977. Survival of bovine enteric bacteria in forest streams and animal wastes. Ellensburg, WA: Central Washington University. 20 p. M.S. thesis.

Doty, R.D.; Hookano, E., Jr. 1974. Water quality of three small watersheds in northern Utah. Res. Note INT–186. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 6 p.

Erlandsen, S.L.; Sherlock, L.A.; Bemrick, W.J. [and others]. 1990. Prevalence of *Giardia spp.* in beaver and muskrat populations in northeastern States and Minnesota: detection of intestinal trophozoites at necropsy provides greater sensitivity than detection of cysts in fecal samples. Applied and Environmental Microbiology. 56: 31–36.

Frost, F.; Plan, B.; Leitchy, B. 1980. *Giardia* prevalence in commercially trapped mammals. Journal of Environmental Health. 42: 245–249.

Geldreich, E.E. 1967. Fecal coliform concepts in stream pollution. Water Sewage Works. 114: R–98 and R–107 to R–110.

Geldreich, E.E. 1970. Applying bacteriological parameters to recreational water quality. Journal of the American Water Works Association. 62: 113–120.

Geldreich, E.E. 1976. Fecal coliform and fecal streptococcus density relationships in waste discharges and receiving waters. Critical Reviews in Environmental Control. 6: 349–369.

Geldreich, E.E.; Kenner, B.A. 1969. Concepts of fecal streptococci in stream pollution. Journal of Water Pollution Control Federation. 41: R336–R352.

Goodrich, Thomas D.; Stuart, David G.; Walter, William G. 1973. Enteric bacteria found in elk droppings in a municipal watershed in Montana. Journal of Environmental Health. 34: 375–377.

Hancock, Dale D.; Besser, Thomas E.; Rice, Daniel H. [and others]. 1998. Multiple sources of *Escherichia coli* O157 in feedlots and dairy farms in the Northwestern U.S.A. Preventive Veterinary Medicine. 35: 11–19.

Jawson, M.D.; Elliott, L.F.; Saxton, K.E.; Fortier, D.H. 1982. The effect of cattle grazing on indicator bacteria in runoff from a Pacific Northwest watershed. Journal of Environmental Quality. 11: 621–627.

Keene, William E.; Sazie, Elizabeth; Kok, Janet [and others]. 1997. An outbreak of *Escherichia coli* O157:H7 infections traced to jerky made from deer meat. Journal of the American Medical Association. 277: 1229–1231.

Kunkle, Samuel H.; Meiman, James R. 1967. Water quality of mountain watersheds. Hydrol. Pap. 21. Ft. Collins, CO: Colorado State University. 52 p.

McFeters, G.A.; Stuart, D.G. 1972. Survival of fecal coliform bacteria in natural waters: field and laboratory studies with membrane-filter chambers. Applied Microbiology. 24: 805–811.

Monzingo, D.L., Jr.; Hibler, C.P. 1987. Prevalence of *Giardia* sp. in a beaver colony and the resulting environmental contamination. Journal of Wildlife Diseases. 23: 576–585.

Moore, A.C.; Herwaldt, B.L.; Craun, G.F. [and others]. 1993. Surveillance for waterborne disease outbreaks—United States 1991–1992. Morbidity and Mortality Weekly Report. 42(SS–5): 1–22.

Niemi, R.M.; Niemi, J.S. 1991. Bacterial pollution of waters in pristine and agricultural lands. Journal of Environmental Quality. 20: 620–627.

Ongerth, J.E.; Hunter, G.D.; DeWalle, F.B. 1995. Watershed use and *Giardia* cyst presence. Water Research. 29: 1295–1299.

Rickard, Lora G.; Siefker, Carla; Boyle, Carolyn R.; Gentz, Edward J. 1999. The prevalence of *Cryptosporidium* and *Giardia* spp. in fecal samples from free-ranging white-tailed deer (Odocoileus virginianus) in the Southeastern United States. Journal of Veterinary Diagnostic Investigation. 11: 65–72.

Riggs, R.A.; Tiedemann, A.R.; Cook, J.G. [and others]. [In press]. Modification of mixed-conifer forests by ruminant herbivores in the Blue Mountains ecological province. Res. Pap. Portland OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Riley, L.W.; Remis, R.S.; Helgerson, S.D. [and others]. 1983. Hemorrhagic colitis associated with a rare *Escherichia coli* serotype. New England Journal of Medicine. 308: 681–685.

Rowland, M.M.; Bryant, L.D.; Johnson, B.K. [and others]. 1997. The Starkey project: history, facilities, and data collection methods for ungulate research. Gen. Tech. Rep. PNW–396: Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 62 p.

Sargeant, Jon M.; Hafer, Dale J.; Gillespie, Jerry R. [and others]. 1999. Prevalence of *Escherichia coli* O157:H7 in white-tailed deer sharing rangeland with cattle. Journal of the American Veterinary Medical Association. 215: 792–794.

Sinton, L.W.; Donnison, A.M.; Hastie, C.M. 1993. Faecal streptococci as faecal pollution indicators: a review. 1: Taxonomy and enumeration. New Zealand Journal of Marine and Freshwater Research. 27: 101–115.

Skinner, Quentin D.; Adams, John C.; Rechard, Paul A.; Beetle, Allen A. 1974. Effect of summer use of a mountain watershed on bacterial water quality. Journal of Environmental Quality. 3: 329–335.

Skinner, Quentin D.; Speck, John E., Jr.; Smith, Michael; Adams, John C. 1984. Stream water quality as influenced by beaver within grazing systems in Wyoming. Journal of Range Management. 37: 142–146.

Stephenson, G.R.; Rychert, R.C. 1982. Bottom sediment: a reservoir of *Escherichia coli* in rangeland streams. Journal of Range Management. 35: 119–123.

Swerdlow, D.L.; Woodruff, B.A.; Brady, R.C. [and others]. 1992. A waterborne outbreak in Missouri of *Escherichia coli* O157:H7 associated with bloody diarrhea and death. Annals of Internal Medicine. 117: 812–819.

Tiedemann, A.R.; Higgins, D.A. 1989. Effects of management strategies on water resources. In: Quigley, T.M.; Sanderson, H.R.; Tiedemann, A.R., eds. Managing interior Northwest rangelands: the Oregon range evaluation project. Gen. Tech. Rep. PNW–238. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 57–91.

Tiedemann, A.R.; Higgins, D.A.; Quigley, T.M. [and others]. 1987. Responses of fecal coliform in streamwater to four grazing strategies. Journal of Range Management. 40: 22–329.

Tiedemann, A.R.; Higgins, D.A.; Quigley, T.M. [and others]. 1988. Bacterial water quality responses to four grazing strategies—comparisons with Oregon standards. Journal of Environmental Quality. 17: 492–498.

Tiedemann, A.R.; Quigley, T.M.; White, L.D. [and others]. 1999. Electronic (fenceless) control of livestock. Res. Pap. PNW–510. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 23 p.

Van Donsel, D.J.; Geldreich, E.E. 1971. Relationships of *Salmonellae* to fecal coliforms in bottom sediments. Water Resources. 5: 1079–1087.

Walter, William G.; Bottman, Robert P. 1967. Microbiological and chemical studies of an open and closed watershed. Journal of Environmental Health. 30: 157–163.

Wells, J.G.; Davis, B.R.; Wachsmuth, I.K. [and others]. 1983. Laboratory investigation of hemorrhagic colitis outbreaks associated with a rare *Escherichia coli* serotype. Journal of Clinical Microbiology. 18: 512–520.

Chapter 16

Water Birds

Christopher A. Nadareski[1]

Introduction

This chapter synthesizes the available information on the sources, mechanisms, and risks of drinking water contamination from water birds. By water birds, we mean geese, ducks, and other duck-like swimming birds such as cormorants, gulls, and wading birds. In general, activities that provide healthy habitat usually help maintain—or at least do not significantly degrade—water quantity and quality for domestic use. However, water birds have been implicated in the contamination of both large and small drinking water supplies.

Issues and Risks

The U.S. Department of Agriculture, Wildlife Services, collect and disseminate information about nuisance-related problems caused by a variety of birds. Birds commonly create human health concerns by polluting potable drinking water supplies and fouling recreation areas, such as swimming beaches. Species like the cormorant are thought to negatively impact recreational fisheries and drinking water supplies. Some terrestrial bird species, such as swallows, pigeons, and starlings, may nest, feed, or roost in water intake structures where their defecation has negative impacts on water quality. Biological, chemical, and nutrient pollutants may be released or deposited by water birds. The type and quantity of pollutant will determine what kind and level of water bird management is required to prevent contamination to a reservoir system.

Estimating impacts from the various types of contamination requires information on the species of water bird inhabiting the water supply, the number of birds per species, the daily defecation or pellet regurgitation rates, the amount of contamination per specified weight of feces or pellet, and the daily amount of activity by each bird on the water. The principal pathogens of concern are listed in table 16.1.

Findings from Studies

Studies have found that water birds and some terrestrial species affect the biological, nutrient, and chemical quality of water.

Contamination

Biological

Among the various types of water bird contaminants are excrement containing bacteria, protozoans, or enteric viruses. These contaminants are routinely monitored by water suppliers and regulated by the U.S. Environmental Protection Agency (EPA) and various State health departments. Numerous studies have documented the occurrence of fecal coliform bacteria and other pathogens in many North American water bird species (Ashendorf and others 1997, Gould and Fletcher 1978, Hatch 1996, Hussong and others 1979). As a result, bird control programs have been developed and implemented at many municipal water sources (Ashendorf and others 1997, Blokpoel and Tessier 1984). Water bird excrement has been reported to contain both human and nonhuman pathogens. Microbiological analysis for human bacterial pathogens found in drinking water are generally represented by the fecal coliform bacteria group which is used as an index to identify the probability one or more of these organisms will be present in the sample (chapter 2 and table 16.1). Fecal coliform originates solely from warm-blooded animals, and analytical methods to identify the origin of bacterial type (human versus nonhuman) are currently under investigation. These new methods require an expensive and time-consuming examination of the sample. As a result, most public water suppliers, at best, can only speculate on the origin of the drinking water bacterial contamination. In the larger municipal water systems, such as New York City's, methods are being developed to link bacterial sources from water samples to human, nonhuman, or both sources.

[1] Research Scientist, New York City Department of Environmental Protection, Valhalla, NY.

Table 16.1—Indicator and principal pathogens of concern in contaminated drinking water

Organism type	Disease	Symptoms
Fecal coliform bacteria group		
Bacteria		
Escherichia coli, Klebsiella pneumonae	Diarrhea, dysentery, hemorrhagic colitis	Diarrhea, nausea, cramps, fever, vomiting, mucus in stools
Important human pathogenic organisms suspected from feces		
Bacteria		
Shigella spp.	Shigellosis	Diarrhea, fever, cramps, tenesmus, blood in stools
Salmonella typhimurium	Salmonellosis	Abdominal pain, diarrhea, nausea, vomiting, fever
S. typhi	Typhoid fever	Abdominal pain, fever, chills, diarrhea or constipation, intestinal hemorrhage
Enterotoxigenic	Diarrhea	Diarrhea, fever, vomiting
Campylobacter jejuni	Gastroenteritis	Abdominal pain suggesting acute appendicitis, fever, headache, malaise, diarrhea, vomiting
Vibrio cholerae (incidence rare or negligible in the United States)	Gastroenteritis	Vomiting, diarrhea, dehydration
Yersinia spp.	Plague, hemorrhagic enterocolitis, terminal ileitis, mesenteric lymphadenitis, septicemia	Diarrhea, fever, abdominal pain
Enteric viruses		
Hepatitis A virus	Hepatitis	Fever, malaise, anorexia, jaundice
Norwalk-like agent	Gastroenteritis	Diarrhea, abdominal cramps, headache, fever, vomiting
Virus-like 27 nm particles	Gastroenteritis	Vomiting, diarrhea, fever
Rotavirus	Gastroenteritis	Vomiting followed by diarrhea for 3 to 8 days
Protozoa		
Giardia lamblia	Giardiasis	Chronic diarrhea, abdominal cramps, flatulence, malodorous stools, fatigue, weight loss
Cryptosporidium	Cryptosporidiosis	Abdominal pain, anorexia, watery diarrhea, weight loss; immuno-compromised individuals may develop chronic diarrhea
Entamoeba histolytica	Amebiasis	Vary from mild diarrhea with blood and mucus to acute or fulminating dysentery with fever and chills

nm particles = nanometers.
Source: Murray and others 1995, Hurst and others 1997.

Nutrients

Contributions of nutrients to aquatic ecosystems by water bird excrement have been well documented (Gould and Fletcher 1978, Manny 1994), but debate continues over the significance of impacts from bird defecation on nutrient loading (Hoyer and Canfield 1994, Murphy 1984). Among the various nutrients identified in water bird excrement, nitrogen and phosphorus are of greatest interest. Both have the potential to increase the rate of eutrophication and degrade the quality of a drinking water supply (Vollenweider and Kerekes 1980).

Chemical

Another type of water bird contamination may be of chemical origin. Glandular releases, body oils, pesticides, and other hydrocarbons are transported by the birds. The impact to drinking water supplies from glandular releases and body oils emitted by water birds is not well understood. Chemical pollutants carried externally on feathers or vectored by water birds, although not well documented, may potentially cause contamination of a water supply. For example, many reservoir systems do not provide adequate food supplies for gulls or other opportunistic feeders. Gulls, therefore, will seek alternative foraging locations, such as agricultural areas, urban centers, moist fields, and landfills, where they may be exposed to a variety of chemicals either through ingestion or external attachment. The gulls may accumulate various chemicals on their feathers and transport them back to the reservoir where they roost. Gulls may also ingest contaminated materials from landfills or sewage treatment facilities, carry them back to the reservoir, and regurgitate them. Additional studies are needed to determine these impacts.

Water Birds as Vectors of Contamination

A variety of human-related activities, such as urbanization, resource exploitation, agriculture, and land conservation, have the potential to promote or discourage populations of a variety of water bird species.

Water birds deposit pollutants in drinking water supplies during roosting, foraging, and overflights. The birds may acquire pathogens, such as *Giardia* spp. or *Cryptosporidium* spp., from domestic farm animals (Graczyk and others 1998) and from urban centers such as shopping malls or landfills. Agricultural operations, such as the spreading of fodder or manure, growing crops, and tilling soil, offer attractive foraging locations for some species that travel great distances. In a study of Canada geese foraging on agricultural land near the Chesapeake Bay, researchers identified a high incidence of *Giardia* and *Cryptosporidium* in their fecal matter (Graczyk 1996) compared to an extremely low incidence of the same two protozoans in geese sampled at an urban reservoir 15 miles north of New York City.

The pathways by which water contaminated with biological materials, chemicals, and nutrients can enter into the reservoir from water birds include streamflow through drainage basins, stormwater surface sheet flow, fly-over fecal releases, and direct fecal deposition. The location of the source of contamination with respect to a reservoir will determine the degree of water-quality impacts. Size and water flow patterns in relation to the water intake structure from which drinking water is drawn also affects the extent of impacts. The extent of contamination depends on the die-off rate, settling rate, and water travel time.

Seasonality of Impacts

Pollutant loadings from water birds vary widely by season. Migratory birds may cause problems only during brief stopovers. Local breeding populations of water birds may also have negative impacts, and those impacts may persist through much of the year. The location of bird activity (roosting, breeding, foraging, etc.) on the reservoir relative to a water intake facility may also determine importance of strategies for managing water bird populations.

Reliability and Limitations of Findings

The types of water-quality impacts are understood. The magnitudes of impacts are less well understood. Resident and migratory water birds will pose similar problems to water supply reservoirs. The magnitude of the impact will vary by species and number of birds inhabiting the reservoir, the surrounding environs and associated sources of contamination, and seasonal patterns of migration. Therefore, water bird impacts will vary and need to be assessed for each reservoir.

Key Points

1. Land managers may have different objectives concerning habitat management for water bird species than reservoir managers. Both forest managers and reservoir managers can and do influence the population dynamics of water bird species. Their management objectives can conflict; for example, the forest manager may want to increase populations of certain water bird species while the reservoir manager may try to decrease or eliminate their activity on or near the reservoir (see New York City case study).
2. To address the contamination of drinking water supplies by water birds, a comprehensive watershed protection plan should be drawn up. This plan needs to identify all major sources of pollution. If it is determined that water birds are major contributors, an additional plan for water bird management should be developed. This water bird management plan should also incorporate objectives both for the land manager and the reservoir manager. Populations of all water bird species that breed or migrate throughout the watershed should be inventoried to identify all potential impacts on water quality. The type of pollution should be identified, as should its potential origins. Where the source of the contaminant is human activity and birds are only a vector, it may be possible to control the contamination by changing the offending human activity.

Case Study: New York City Waterfowl Management Program

As the Nation's largest, unfiltered water supplier, the city of New York Department of Environmental Protection (DEP) is responsible for the maintenance of 19 reservoirs, 3 controlled lakes encompassing an area of almost 2,000 square miles, and serving 9 million consumers. Beginning in the early 1980's, the city enhanced its water-quality monitoring programs to address watershed protection issues. As more stringent Federal regulations were implemented through the Safe Drinking Water Act of 1986 and subsequent Surface Water Treatment Rule of 1989, New York City remained determined to maintain its unfiltered status through stringent criteria set by the EPA. In order to fulfill this filtration avoidance determination, the DEP developed a comprehensive Watershed Protection Program (WPP) to identify and eliminate sources of pollution that would compromise its water quality. An important component of the WPP was the implementation of a Waterfowl Management Program that identified birds, particularly gulls, geese, ducks, and cormorants, as the primary source of fecal coliform bacteria to the water system. Baseline bird population data was well correlated with the seasonal elevations of bacteria. As a result, DEP instituted a bird deterrent/harassment program to eliminate the presence of both breeding and migratory Canada geese and migratory gulls, ducks, and other water birds. Techniques used to deter the birds included shoreline fencing, meadow management to deter feeding opportunities, bird distress tapes, and the use of pyrotechnics from motorboats and hovercraft from dawn to dusk on a daily basis. The techniques were highly effective: elimination of defecation from roosting birds eliminated the seasonal elevations of bacteria, allowing New York City to maintain its filtration avoidance status.

Literature Cited

Ashendorff, A.; Principe, M.A.; Seeley, A. [and others]. 1997. Watershed protection for New York City's supply. Journal of the American Water Works Association. 89(3): 75–88.

Blokpoel, G.; Tessier, G.D. 1984. Overhead wires and monofilament lines exclude ring-billed gulls from public places. Wildlife Society Bulletin. 12: 55–58.

Gould, D.J.; Fletcher, M.R. 1978. Gull droppings and their effects on water quality. Water Research. 12: 665–672.

Graczyk, T.K.; Fayer, R.; Trout, J.M. [and others]. 1998. *Giardia* sp. cysts and infectious *Cryptosporidium parvum* oocysts in the feces of migratory Canada geese *(Branta canadensis)*. Applied and Environmental Microbiology. 64(7): 2736–2738.

Hatch, Jeremy J. 1996. Threats to public health from gulls *(Laridae)*. International Journal of Environmental Health Research. 6: 5–16.

Hoyer M.V.; Canfield, D.E. 1994. Bird abundance and species richness on Florida lakes—influence of trophic status, lake morphology, and aquatic macrophytes. Hydrobiologia. 280: 107–120.

Hurst, C.J.; Knudsen, G.R.; McIvering, M.J. [and others]. 1997. Manual of environmental microbiology. Washington, DC: American Society for Microbiology Press. 894 p.

Hussong, D.; Damare, J.M.; Limpert, R.J. [and others]. 1979. Microbial impact of Canada geese (*Branta canadensis*) and whistling swans *(Cygnus columbianus columbianus*) on aquatic ecosystems. Applied and Environmental Microbiology. 37(1): 14–20.

Manny, B.A.; Johnson, W.C.; Wetzel, R.G. 1994. Nutrient additives by waterfowl to lakes and reservoirs: predicting their effects on productivity and water quality. Hydrobiologia. 279: 121–132.

Murphy, S.M.; Kessel, B.; Vining, L.J. 1984. Waterfowl populations and limnological characteristics of taiga ponds. Journal of Wildlife Management. 48(4): 1156–1163.

Murray, P.R.; Baron, E.J.; Pfaller, M.A. [and others]. 1995. Manual of clinical microbiology. 6th ed. Washington, DC: American Society for Microbiology Press. 1,482 p.

New York City Department of Environmental Protection. 1993. Implications of phosphorus loading for water quality in NYC Reservoirs. Corona, NY: New York City Department of Environmental Protection. 65 p.

U.S. EPA. 1999. U.S. EPA's unfiltered water supply systems guidance manual—revised draft. Washington, DC: U.S. Environmental Protection Agency. 58 p.

Vollenweider, R.A.; Kerekes, J.J. 1980. Background and summary results of the OECD cooperative program on eutrophication. EPA 440/5–81–010:25–36. Washington, DC: Environmental Protection Agency: 25–35.

Chapter 17

Fish and Aquatic Organisms

C. Andrew Dolloff [1]

Introduction

Freshwater fish management can impact water quality by manipulating fish and other organisms and by altering the physical, chemical, or biological attributes of habitat. Fishery managers manipulate populations for a variety of reasons, including attempts to maximize yield, control size and age structure, establish populations of desirable and remove populations of undesirable species, and to reestablish populations and communities that have been extirpated. Habitat management often accompanies attempts to manipulate populations. All components of habitat are subject to manipulation, including not only the obvious structural elements, such as substrate, cover, and flow obstructions, but also water chemistry and the content of the biological community.

Fish Hatcheries and Aquaculture Facilities

Issues and Risks

Fish hatcheries or fish culture facilities can reduce water quality. Untreated effluent from these facilities typically consists of metabolic waste products and solids derived from uneaten fish food and fish wastes (Goldburg and Triplett 1997). In addition, culture facilities also may intermittently discharge pathogenic bacteria and parasites and the chemicals and drugs used in the prevention or treatment of disease (Liao 1970, Piper and others 1986). The usual waste stream from a culture facility can be treated as a chronic point source and effluent quality and quantity can be monitored at or immediately below the outfall. Monitoring results tend to be highly variable because of variation in production schedules and other activities (Foy and Rosell 1991) such as periodic cleaning and flushing (Bergheim and others 1984).

Findings from Studies

The quality of effluent water varies considerably depending on the specific features of the culture system including the species of fish, intensity of production, the diet and feeding regime, and the temperature and chemical character of source water (Axler and others 1997). The four most important dissolved constituents of fish cultural wastewater include ammonia, nitrate, phosphate, and organic matter. Ammonia is toxic to most aquatic life and all four constituents are primary agents of eutrophication. Along with dissolved organic matter, suspended solids, which are predominantly organic, contribute to biological oxygen demand in receiving waters. Nutrient loading can be significant, as 60 to 75 percent of the nitrogen and phosphorus in fish food ultimately becomes part of the waste stream (Axler and others 1997). Unless intakes for domestic water supplies are located immediately downstream from the outfall, however, hatchery effluents are not likely to severely degrade domestic water supplies.

Effluents from trout farms may diminish water quality slightly during periods of low flow and high temperature (Selong and Helfrich 1998). In a study of five Virginia trout farms, Selong and Helfrich (1998) found that total ammonia nitrogen, un-ionized ammonia nitrogen, and nitrite nitrogen levels increased downstream from effluent outfalls but did not exceed thresholds for lethal exposure for aquatic organisms. Dissolved oxygen levels also decreased but were typically above 7.0 parts per million (ppm). Temperature, pH, nitrate nitrogen, and total phosphorus levels did not differ from upstream levels. Substrate embeddedness was greater below outfalls from two farms but settleable solids concentrations were always <0.1 ppm. The lack of significant water-quality degradation reflected the tendency of growers to adjust production to correspond to periods of high flow and low-to-moderate temperature (Selong and Helfrich 1998). In Washington State, however, hatchery effluents during the summer had significantly elevated temperature, pH, suspended solids, ammonia, organic nitrogen, total phosphorus, and biological oxygen demand (Kendra 1991). Phosphorus loading in hatchery effluent in one Washington State creek was equivalent to secondarily treated sewage discharge from a town of 2,300 people. The

[1] Project Leader, USDA Forest Service, Southern Research Station, Blacksburg, VA.

influence of hatchery effluents tends to be localized in the immediate downstream reach (Doughty and McPhail 1995). Effects on periphyton and macroinvertebrates were not detected about 1,200 feet [366 meters (m)] downstream of the outfalls from five Virginia trout farms (Selong and Helfrich 1998).

Because most of the total nutrient load produced at a typical facility is in the form of settleable solids, treatment usually consists of diverting wastewater through settling basins before discharge (Piper and others 1986). Much of the variation observed in effluent waste among facilities can be traced to differences in settling characteristics (Mudrak 1981). When located near towns or cities, culture facilities can avoid discharging directly into streams or rivers by connecting to municipal sewage treatment facilities.

Fish disease organisms, chemicals, and other additives commonly used in culture facilities also may influence the quality of water for domestic use. Fish trematodes, cestodes, and nematodes can be transmitted to people who eat certain species of raw fish. In general, however, fish diseases do not present risks to human health (Hoffman 1999). Most potential problems with fish disease and water quality arise from the chemicals and procedures used in treatment. At present, relatively few chemicals have been approved by the U.S. Food and Drug Administration (FDA) for use in the treatment of fish diseases. The antibacterials approved for use on food fishes include oxytetracycline, sulfamethoxine (Romet), and sulfamerazine. Most drugs are administered through feed, from which a considerable fraction may be released into the environment. Release occurs through three routes: uningested food, feces, and in urine and bile fluid (Capone and others 1996). Residues of oxytetracycline in sediments under net pens of intensively cultured (and treated) Atlantic salmon were present for at least 10 months after treatment (Capone and others 1996). Other drugs approved for use in treating fish include formalin (for the treatment of external parasites and fungus on fish eggs) and tricaine methanesulfonate, an anesthetic. The FDA also maintains a list of unapproved new animal drugs (drugs that have not gone through the formal approval process but are not expected to have a negative impact on the environment) for use in aquaculture of food fishes provided the following conditions are met:

1. The drugs are used for the prescribed indications, including species and life stage;
2. The drugs are used at the prescribed dosages;
3. The drugs are used according to good management practices;
4. The product is of an appropriate grade for use in food animals; and
5. An adverse effect on the environment is unlikely.

Among a host of common compounds and substances in this classification are sodium chloride and ice. Examples of unapproved drugs for use in aquaculture are acetic acid (parasiticide for fish), calcium chloride (ensures proper egg hardening, aids in maintaining osmotic balance in fish), carbon dioxide gas (fish anesthetic), hydrogen peroxide (fungicide), magnesium sulfate (controls monogenetic trematode and external crustacean infestations), potassium chloride (relieves osmoregulatory stress and prevents shock), povidone iodine compounds (fish egg disinfectant), sodium sulfate (improves egg hatchability), and urea and tannic acid (denatures adhesive component of fish eggs).

In addition to treating specific diseases, all culture facilities must undergo periodic cleaning and sterilization. Chlorine (HTH) is often used for this purpose; exposure to 5 ppm for 1 hour kills nearly everything. Chlorine rapidly loses its toxicity (1 day or less) and can be neutralized by sodium thiosulfate.

Reliability and Limitations of Findings

A large body of research has accumulated on the design, construction, operation, and maintenance of fish culture facilities. Because a significant portion of this research has addressed the issue of effluent management, the findings are considered highly reliable. These findings are widely applicable to flowing water.

Research Needs

None identified.

Key Points

Existing knowledge will permit managers and policymakers to make informed decisions about the impact of culture facilities on domestic water supplies.

Chemical Reclamation

Issues and Risks

Piscicidal (fish-killing) compounds have been used to sample, control, or eradicate fish populations since the 1930's (Bettoli and Maceina 1996). Purposes have included eradication of exotic species such as common carp from ponds, lakes, and reservoirs and removal of nonnative species from headwater streams. With the exception of sampling programs, the objective of most piscicide applications is to remove one or more species so that the water body can be stocked soon after poisoning with species considered more desirable.

Findings from Studies

Only registered piscicides may be applied to water in North America. Among a variety of possible compounds, only rotenone (trade name Noxfish), antimycin A (trade name Fintrol), 3-trifluoromethyl-4-nitrophenol (TFM), and Bayluscide (a nitosalicylanilide salt, trade name Bayer 73) are approved for use in the United States. Bayluscide and TFM, either alone or in combination, are used exclusively to sample or control sea lamprey, primarily in streams flowing into the Great Lakes (Bettoli and Maceina 1996, Marking 1992). In the United States, lampricides can be applied only by personnel certified by the U.S. Fish and Wildlife Service or an approved State conservation agency (Bettoli and Maceina 1996).

Rotenone, a naturally occurring chemical derived from the roots of tropical plants in the genera *Derris* and *Lonchocarpus*, has been used for centuries by native people to kill fish for food. Rotenone is highly toxic to fish, which are killed by disruptions in cellular respiration (Haley 1978). Most nontarget aquatic organisms usually are not affected by the concentrations of rotenone used to kill fish, but high doses can kill amphibians, some reptile species, and a variety of macroinvertebrates (Bettoli and Maceina 1996). The toxicity and persistence of rotenone are influenced by turbidity, temperature, and pH; in general, rotenone is most toxic in clear, warm, acidic water. Residues of rotenone were detectable in the bottom sediments of experimental ponds in Wisconsin for nearly 14 days during the spring at 46 °F (8 °C), but decreased below the limits of detection within 3 days during the summer at 72 °F (22 °C) and during the fall at 59 °F (15 °C) (Dawson and others 1991). Higher temperatures coupled with the presence of clay (bentonite) led to adsorption and rapid decline in residues. Dawson and others (1991) suggested that at warmer temperatures, water treated with rotenone would be safe for swimming immediately after treatment with a concentration of 250 parts per billion (ppb). A strong oxidizing agent such as potassium permanganate (applied at 2.0 to 2.5 ppm) will detoxify rotenone when applied downstream of treatment sections in flowing water or in lakes where managers wish to limit the area of kill.

Rotenone has a low mammalian toxicity (Marking 1988) and relatively short half-life (<1 day in water at 73 °F) (Gilderhus and others 1988). Nevertheless, negative public perceptions about the use of any poisonous substance in open water have curtailed the use of rotenone over the last decade. Older formulations of rotenone, no longer approved for sale in the United States, sometimes contained carrier substances such as trichloroethylene and piperonyl butoxide, which are known carcinogens. Residues of the latter were detected up to 9 months after treatment of Lake Davis in California and caused disruption of a public drinking water supply that drew national publicity.

Similar to rotenone, antimycin is relatively nontoxic to mammals (Herr and others 1967) but highly poisonous to fish. Antimycin toxicity in water varies by species, and concentrations over 100 ppb may be required to kill resistant species (Bettoli and Maceina 1996). The effectiveness of antimycin may decrease below 50 °F (10 °C) (Tiffan and Bergersen 1996), but in warm water, temperature does not greatly influence its toxicity. High turbidity (Gilderhus 1982) and alkalinity (Tiffan and Bergersen 1996) decrease its effectiveness and persistence (Bettoli and Maceina 1996, Lee and others 1971). Turbulence leading to oxidation and foaming also may contribute to antimycin inactivation (Tiffan and Bergersen 1996).

Reliability and Limitations of Findings

A large body of research has accumulated on the short- and long-term effects of chemical reclamation. Studies have been conducted in both warmwater and coldwater habitats under widely varying conditions. The findings are considered highly reliable and widely applicable.

Research Needs

None identified.

Key Points

Existing knowledge will permit managers and policymakers to estimate impacts of piscicides on fish populations and water quality. However, controversy likely will continue over the use of piscicides in sources of public water supplies.

Restoration and Reintroduction of Populations and Communities

Issues and Risks

With the growth in environmental awareness in recent years, management agencies increasingly are called upon to restore species that have been intentionally or accidentally extirpated from aquatic ecosystems. In most situations, the impact of the restoration on water quality will be almost entirely positive because improved water quality is one of the prerequisites for survival of target organisms. Managers must be aware, however, that one of the most obvious signs of a successful restoration—large numbers of fish—may contribute to periodic, temporary declines in water quality. Large numbers of fish may cause problems with water quality during periods of low flow and high temperature if the fish become stressed, die, and decompose. Large numbers of fish of a single species seldom die in a small area under such conditions except when anadromous fish such as Pacific salmon or alosids (shads and alewifes) congregate for spawning.

Findings from Studies

Although some aspects of water quality may be temporarily degraded by the decay of anadromous fish carcasses, the long-term effect can have great ecological value. Nutrients from anadromous fish carcasses help maintain the productive capacity of streams and riparian zones in coastal watersheds of the Pacific Northwest and Alaska (Cederholm and others 1989, 1999; Wilson and Halupka 1995) and streams of the Atlantic Coastal Plain (Garman and Macko 1998). On the west coast, the carcasses of abundant pink (Brickell and Goering 1970) and sockeye salmon (Kline and others 1994) and the less abundant coho salmon (Bilby and others 1996) contribute important nutrients, particularly nitrogen, to otherwise nutrient-poor watersheds. Similar relationships have been observed in Atlantic slope drainages (Durbin and others 1979, Garman 1992).

Accidental introductions generally are of greater concern both ecologically and for water quality. Perhaps the most troublesome, accidentally introduced aquatic pest is the zebra mussel *Dreissena polymorpha*. This bivalve mollusk first arrived in North America in ship's ballast via the St. Lawrence Seaway sometime around 1986 (Hebert and others 1989). Since that time, it has invaded all of the Great Lakes and the major rivers of the Eastern United States (Ludyanskiy and others 1993). Invasion of virtually all the major river systems in North America is viewed by some as inevitable, with only the specific timetable subject to question (Morton 1997). Zebra mussels will attach to virtually any surface, including the interior of water inlet pipes. Zebra mussels also will attach to the hulls of both large, commercial and smaller recreational vessels, which act as dispersal agents (Keevin and others 1992). The primary economic cost of the zebra mussel invasion has been the fouling of intakes for raw industrial and potable water. Zebra mussels may be present in numbers sufficient to clog water intakes, necessitating either abandonment or laborious and repeated cleaning of the pipes. Over $4 billion per year may ultimately be spent on attempts to control or mitigate zebra mussel impacts (Morton 1997). The ecological impacts of the invasion have been equally profound, particularly on native unionid mussels (Haag and others 1993). Nalepa and Schloesser (1993) completely reviewed issues surrounding the zebra mussel invasion.

The accidental introduction of the zebra mussel, while certainly regrettable, may have future benefits. Morton (1997) noted that in Europe, where the species has been established for around 170 years, zebra mussels are used as biomonitors for trace metals and radionuclides. Their natural water filtering abilities have made them useful both for restoration of natural water systems and in the treatment of human sewage. With development of genetically sterile stocks, it may be possible to employ zebra mussels or other suitable species in the cleanup of both natural and artificial water bodies in North America (Morton 1997).

Reliability and Limitations of Findings

Although this area of research and restoration is relatively recent, the findings are generally considered reliable. Restoration ecology is one of the newest branches of ecology. Relatively little research has accumulated and there are few long-term studies of the effects of reintroduced or accidentally introduced species. The likely impacts of exotic invasive species, such as the zebra mussel, on domestic water supplies, however, can be predicted with a high degree of precision.

Research Needs

1. Long-term studies are needed on the impacts of both reintroduced species and exotic species on habitat and water quality.
2. Additional research should address the losses of other species and reductions in water quality after introduction of exotic or previously extirpated species.

Key Points

The current body of research will permit managers to address some issues, but the long-term influence on water quality of repeated measures to control exotic species is unknown.

Physical Habitat

Issues and Risks

The literature on stream habitat improvement is large and diverse, ranging from simple handbooks and pamphlets designed for volunteers to more detailed treatments for biologists and other professionals, e.g., Hunter (1991). Most habitat improvements are designed to slow or redirect water flow or to create pools. Habitat improvements include engineered structures, such as k-dams, wing dams, and deflectors. Unless they are constructed with chemically preserved materials (such as creosote or pressure-treated wood), the materials used in most structures should not pose a direct threat to water quality. Indirectly, structures may cause changes in sediment storage and routing patterns by causing excessive scouring of channel bottoms and sides. Turbidity may increase after installation of even properly sited structures, but structures installed without regard for natural channel processes can cause major disruptions in flow patterns and trigger accelerated channel erosion.

Only recently have managers attempted to mimic the natural structure of streams by adding native materials such as large or coarse woody debris (CWD) to streams. For purposes of scientific discussion, CWD includes any piece of wood that is at least 4 inches [10 centimeters (cm)] in diameter and 3 to 4 feet (0.9 to 1.2 m) long (Bisson and others 1987, Dolloff 1994, Harmon and others 1986, Maser and Sedell 1994). In practice or application, managers typically consider CWD to be wood that is at least 12 inches (31 cm) in diameter with length equal to the width of the receiving stream channel. Logging residue or slash is excluded from most habitat enhancement projects because it tends to be unstable in all but the smallest stream channels. Woody debris enters stream channels naturally by a number of routes, including bank undercutting, windthrow, and as a result of catastrophic events, such as snow and debris avalanches and hurricanes. Managers add CWD by direct felling or toppling of streamside trees or by transport from more distant sources.

Findings from Studies

In general, the relatively small amounts of wood added by managers to enhance or restore stream habitat are not likely to exert a major impact on water quality. There can be exceptions when pieces dislodged during floods plug culverts, bridge openings, or other structures and cause accelerated erosion or the failure of streamside roads or road crossings. When large amounts of fine debris from bark and branches accumulate, dissolved oxygen may be depleted and hydrogen sulfide and ammonia produced (Sedell and others 1991). Leachates from logs may contain toxins, but they are unlikely to reach significant concentrations under natural conditions (Schaumburg 1973, Thut and Schmeige 1991).

Reliability and Limitations of Findings

Since intensive research began about 20 years ago, a large body of information has accumulated on the ecology and management of CWD in streams. Many studies, including historical observations and experiments, have been conducted in watersheds across North America and several other continents. Reliability is high.

Studies have demonstrated the benefits of large wood in streams. Whether in small streams or larger rivers, research suggests that wood in the water is good for fish and, except in certain well-defined situations, not detrimental to domestic water supplies.

Research Need

Research is needed to evaluate the influence of woody vegetation planted to stabilize fish habitat. For example, use of black locust or other nitrogen-fixing plants may increase nitrogen content of the water, resulting in either excessive algal blooms or the need to remove the nitrogen prior to domestic use.

Key Point

The information available should allow managers to address most issues related to the effects on drinking water of woody materials installed to improve fish habitat.

Liming of Acidified Waters

Issues and Risks

Water from areas where the bedrock does not have a high buffering capacity can be acidified by major soil disturbing activities such as road building, mining (Nelson and others 1991), or acid precipitation. Associated problems include disruption of physiological processes for many aquatic organisms and increased concentrations of toxic forms of metals such as aluminum. Left uncorrected, continuing acidification can kill entire faunas (Olem 1991). The quality of surface water can be dramatically improved by treating streams, lakes, or whole catchments with a soluble basic mineral such as crushed limestone.

Findings from Studies

The beneficial effects of liming have been known for hundreds of years (Henrikson and Brodin 1995, Porcella and others 1995). Liming, however, is at best a temporary solution and must be repeated to maintain water quality. The primary benefit of liming has been to preserve or recover fish stocks and diversity of aquatic species. Liming generally increases pH and reduces concentrations of metals, including toxic forms of aluminum (Wilander and others 1995). Concentrations of many other trace metals also decreases as pH increases (Vesely 1992). Liming may cause the precipitation of metals and stress to aquatic species present in mixing zones, such as in downstream reaches where acidic tributaries join with limed water (Henrikson and Brodin 1995). The effects of liming are nearly always positive; however, limestone sand or gravel applied directly to a stream may contribute to the sediment load during high flows.

Reliability and Limitations of Findings

In the last 20 to 30 years, much research has described the mechanism, consequences, and treatment of acidified surface water. The findings are considered highly reliable.

Impacts of liming on water quality have often been studied, and the findings are widely applicable.

Research Needs

In general, knowledge of virtually all of the long-term (more than 20 years) effects of liming on aquatic ecosystems and water quality is lacking. Henrikson and Brodin (1995) compiled a comprehensive list of research questions about liming. In the area of water quality, some of the topics needing particular attention include the effects of liming on the uptake and transport of mercury and other metals in aquatic organisms and the consequences of reacidification if liming is stopped.

Key Point

For treatment of individual small watersheds, current knowledge should permit managers to address most issues related to sources of drinking water associated with liming.

Literature Cited

Axler, R.P.; Tikkanen, C.; Henneck, J. [and others]. 1997. Characteristics of effluent and sludge from two commercial rainbow trout farms in Minnesota. The Progressive Fish-Culturist. 59: 161–172.

Bergheim, A.; Hustveit, H.; Kittelsen, A.; Selmer-Olsen, A.R. 1984. Estimated pollution loadings from Norwegian fish farms. 2: Investigations 1980–1981. Aquaculture. 36: 157–168.

Bettoli, P.W.; Maceina, M.J. 1996. Sampling with toxicants. 2d ed. In: Murphy, B.R.; Willis, D.W., eds. Fisheries techniques. Bethesda, MD: American Fisheries Society. 732 p. Chapter 10.

Bilby, R.E.; Fransen, B.R.; Bisson, P.A. 1996. Incorporation of nitrogen and carbon from spawning coho salmon into the trophic system of small streams: evidence from stable isotopes. Canadian Journal of Fisheries and Aquatic Sciences. 53: 164–173.

Bisson, P.A.; Bilby, R.E.; Bryant, M.D. [and others]. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. In: Salo, E.O.; Cundy, T.W., eds. Streamside management: forestry and fishery interactions. Seattle: University of Washington: 143–190.

Brickell, D.C.; Goering, J.J. 1970. Chemical effects of salmon decomposition on aquatic ecosystems. In: Murphy, R.S.; Nyquist, D., eds. First international symposium on water pollution control in cold climates; [Date of meeting unknown]; Fairbanks, AK.Washington, DC: U.S. Government Printing Office: 125–138.

Capone, D.G.; Weston, D.P.; Miller, V.; Shoemaker, C. 1996. Antibacterial residues in marine sediments and invertebrates following chemotherapy in aquaculture. Aquaculture. 145: 55–75.

Cederholm, C.J.; Houston, D.B.; Cole, D.L.; Scarlett, W.J. 1989. Fate of coho salmon carcasses in spawning streams. Canadian Journal of Fisheries and Aquatic Sciences. 46: 1347–1355.

Cederholm, C.J.; Kunze, M.D.; Murota, T.; Sibatani, A. 1999. Pacific salmon carcasses: essential contributions of nutrients and energy for aquatic and terrestrial ecosystems. Fisheries. 24(10): 6–15.

Dawson, V.K.; Gingerich, W.H.; Davis, R.A.; Gilderhus, P.A. 1991. Rotenone persistence in freshwater ponds: effects of temperature and sediment adsorption. North American Journal of Fisheries Management. 11: 226–231.

Dolloff, C.A. 1994. Large woody debris: the common denominator for integrated environmental management of forest streams. In: Cairns, J.; Crawford, T.V.; Salwasser, H., eds. Implementing integrated environmental management. Blacksburg, VA: Virginia Tech Press: 93–108.

Doughty, C.R.; McPhail, C.D. 1995. Monitoring the environmental impacts and consent compliance of freshwater fish farms. Aquaculture Research. 26: 557–565.

Durbin, A.G.; Nixon, S.W.; Oviatt, C.A. 1979. Effects of the spawning migration of the alewife *Alsoa pseudoharengus* on freshwater ecosystems. Ecology. 60: 8–17.

Foy, R.H.; Rosell, R. 1991. Loadings of nitrogen and phosphorus from a Northern Ireland fish farm. Aquaculture. 96: 17–30.

Garman, G.C. 1992. Fate and significance of postspawn clupeid fish carcasses in a large mid-Atlantic river. Transactions of the American Fisheries Society. 121: 123–132.

Garman, G.C.; Macko, S.A. 1998. Contribution of marine-derived organic matter to an Atlantic coast, freshwater, tidal stream by anadromous clupeid fishes. Journal of the North American Benthological Society. 17: 277–285.

Gilderhus, P.A. 1982. Effects of an aquatic plant and suspended clay on the activity of fish toxicants. North American Journal of Fisheries Management. 2: 301–306.

Gilderhus, P.A.; Dawson, V.K.; Allen, J.L. 1988. Deposition and persistence of rotenone in shallow ponds during cold and warm seasons. Invest. in fish control 95. Washington, DC: U.S. Fish and Wildlife Service. 7 p.

Goldburg, R.; Triplett, T. 1997. Murky waters: environmental effects of aquaculture in the U.S.: executive summary and preface. New York: Environmental Defense Fund. 21 p.

Haag, W.R.; Berg, D.J.; Garton, D.W.; Farris, J.L. 1993. Reduced survival and fitness in native bivalves in response to fouling by the introduced zebra mussel (*Dreissena polymorpha*) in western Lake Erie. Canadian Journal of Fisheries and Aquatic Sciences. 50: 13–19.

Haley, T.J. 1978. A review of the literature on rotenone 1,2,12,12a-tetrahydro–8,9–dimethoxy–2–(1–methylethenyl)–1–benzopyrano [3,5–b]furo[2,3–h][1]benzopyran–6(6h)–one. Journal of Environmental Pathology and Toxicology. 1: 315–337.

Harmon, M.E.; Franklin, J.E.; Swanson, F.J. [and others]. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research. 15: 133–302.

Hebert, P.D.N.; Muncaster, B.W.; Mackie, G.L. 1989. Ecological and genetic studies on *Dreissena polymorpha* (Pallis): a new mollusc in the Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences. 46: 1587–1591.

Henrikson, L.; Brodin, Y.W. 1995. Liming of surface waters in Sweden—a synthesis. In: Henrikson, L.; Brodin, L.W., eds. Liming of acidified surface waters—a Swedish synthesis. Berlin, Germany: Springer-Verlag: 1–44

Herr, F.E.; Greslin, E.; Chappel, C. 1967. Toxicological studies of antimycin, a fish eradicant. Transactions of the American Fisheries Society. 96: 320–326.

Hoffman, G.L. 1999. Parasites of North American freshwater fishes. 2[d] ed. Ithaca, NY: Comstock Publishers. 539 p.

Hunter, C.J. 1991. Better trout habitat—a guide to stream habitat restoration and management. Washington, DC: Island Press. 320 p.

Keevin, T.M.; Yarbrough, R.E.; Miller, A.C. 1992. Long-distance dispersal of zebra mussels (*Dreissena polymorpha*) attached to hulls of commercial vessels. Journal of Freshwater Ecology. 7: 437.

Kendra, W. 1991. Quality of salmonid hatchery effluents during a summer low-flow season. Transactions of the American Fisheries Society. 120: 43–51.

Kline, T.C., Jr.; Goering, J.J.; Mathisen, O.A. [and others]. 1994. Recycling of elements transported upstream by runs of Pacific salmon: 2. ^{15}N ^{13}C evidence in the Kvichak River watershed, Bristol Bay, southwest Alaska. Canadian Journal of Fisheries and Aquatic Sciences. 50: 2350–2365.

Lee, T.H.; Derse, P.H.; Morton, S.D. 1971. Effects of physical and chemical conditions on the detoxification of antimycin. Transactions of the American Fisheries Society. 100: 13–17.

Liao, P.B. 1970. Potential pollution of salmonid fish hatcheries. Water Sewage Works. 117: 291–297.

Ludyanskiy, M.L.; McDonald, D.; MacNeill, D. 1993. Impact of the Zebra mussel, a bivalve invader. Bioscience. 43: 533–544.

Marking, L.L. 1988. Oral toxicity of rotenone to mammals. Invest. in fish control 94. Washington, DC: U.S. Fish and Wildlife Service. 5 p.

Marking, L.L. 1992. Evaluation of toxicants for the control of carp and other nuisance fishes. Fisheries. 17: 6–12.

Maser, C.; Sedell, J. 1994. From the forest to the sea: the ecology of wood in streams, rivers, estuaries, and oceans. Delray Beach, FL: St. Lucie Press. 195 p.

Morton, B.M. 1997. The aquatic nuisance species problem: a global perspective and review. In: D'Itri, F.M., ed. Zebra mussels and aquatic nuisance species. Chelsea, MI: Ann Arbor Press, Inc.: 1–54.

Mudrak, V.A. 1981. Guidelines for economical commercial fish hatchery wastewater treatment systems. In: Allen, L.J.; Kinney, E.C., eds. Proceedings of the bioengineering symposium for fish culture; [Date of meeting unknown]; Traverse City, MI. Bethesda, MD: American Fisheries Society, Fish Culture Section: 174–182.

Nalepa, T.F.; Schloesser, D.W., eds. 1993. Zebra mussels: biology, impacts, and control. Boca Raton, FL: Lewis Publishers. 810 p.

Nelson, R.L.; McHenry, M.L.; Platts, W.S. 1991. Mining. In: Meehan, W.R., ed. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Soc. Spec. Publ. 19. Bethesda, MD: American Fisheries Society. 751 p. Chapter 12.

Olem, H. 1991. Liming acidic surface waters. Chelsea, MI: Lewis Publishers. 331 p.

Piper, R.G.; McElwain, I.B.; Orme, L.E. [and others]. 1986. Fish hatchery management. Bethesda, MD: American Fisheries Society. 517 p. In cooperation with: U.S. Fish and Wildlife Service. [Reprinted with corrections from the 1982 original].

Porcella, D.B.; Driscoll, C.T.; Schofield, C.L.; Newton, R.M. 1995. Lake and watershed neutralization strategies. Journal of Water, Air and Soil Pollution. 85: 889–894.

Schaumburg, F.D. 1973. The influence of log handling on water quality. EPA–R2–73–085. Washington, DC: U. S. Environmental Protection Agency. 105 p.

Sedell, J.R.; Leone, F.N.; Duval, S. 1991. Water transportation and storage of logs. In: Meehan, W.R., ed. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Soc. Spec. Publ. 19. Bethesda, MD: American Fisheries Society. 751 p. Chapter 9.

Selong, J.H.; Helfrich, L.A. 1998. Impacts of trout culture effluent on water quality and biotic communities in Virginia headwater streams. The Progressive Fish-Culturist. 60: 247–262.

Tiffan, K.F.; Bergersen, E.P. 1996. Performance of antimycin in high-gradient streams. North American Journal of Fisheries Management. 16: 465–468.

Thut, R.N.; Schmiege, D.C. 1991. Processing mills. In: Meehan, W.R., ed. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Soc. Spec. Publ. 19. Bethesda, MD: American Fisheries Society. 751 p. Chapter 10.

Veselý, J. 1992. Effects of acidification on trace metal transport in fresh waters. In: Steinberg, C.E.W.; Wright, R.F., eds. Acidification of freshwater ecosystems: implications for the future. Cichester, England: John Wiley: 141–151.

Wilander, A.; Andersson, P.; Borg, H.; Broberg, O. 1995. The effects of liming on water chemistry. In: Henrikson, L.; Brodin, Y.W., eds. Liming of acidified surface waters—a Swedish synthesis. Berlin, Germany: Springer-Verlag. 458 p. Chapter 5.

Wilson, M.; Halupka, K. 1995. Anadromous fish as keystone species in vertebrate communities. Conservation Biology. 9: 489–497.

Part V:

Effects of Mining and Oil and Gas Development on Water Quality

Greens Creek Mine, Tongass National Forest, Alaska, with sediment pond.
Photo by Stephen Glasser

Chapter 18

Hardrock Mining

Mike Wireman[1]

Introduction

Mining can significantly impact the quality of water used for domestic and municipal water supplies. These impacts can be brief or long lasting, and they differ with the type of ore, the mining method, the method of ore processing, the effectiveness of water management, and after mining ceases, the overall nature of mine closure. The impacts include transport and deposition of sediment, acid runoff, and release and transport of dissolved metals and other associated mine contaminants.

Hardrock mining is defined as the extraction of precious and industrial metals and nonfuel minerals by surface and underground mining methods (Lyon and others 1993). In the United States, extensive hardrock mining started in the 1880's, and, for the next 70 to 80 years, was a major industry in many States. Many metals and minerals produced by hardrock mining are valuable natural resources and have been important to the economy of many States. The legacy of the active period of hardrock mining includes more than 200,000 abandoned or inactive mines. As of 1992, there were more than 500 operating mines in the United States, of which, more than 200 are gold mines. As of 1997, there were approximately 60 mine sites in 26 States on the Federal Superfund National Priorities List because of serious pollution problems.

Hardrock mining is a large-scale activity that typically disturbs large areas of land. Unlike other industrial facilities, mines must be located at specific places where ore bodies are found. Many ore bodies and mines are located on public land administered by Federal land management agencies—the Forest Service in the U.S. Department of Agriculture and several agencies in the U.S. Department of the Interior. Mines on public land are frequently located in watersheds with relatively little development. Unless proper environmental controls are used during mining and ore processing, and after mine closure, serious environmental damage can result. During the first half of the 20th century, environmental controls were very limited or nonexistent and, as a result, numerous abandoned mines continue to cause serious environmental damage. Ownership of abandoned mines on public land is often difficult or impossible to establish. To date, the Forest Service does not have a complete inventory of these mines. However, some State mine-permitting agencies have compiled inventories.

Because of the high waste-to-product ratios associated with mining most ore bodies, large volumes of waste are generated. Mine waste includes all of the leftover material generated as a result of mining and ore processing activities. Most mine waste is considered to be nonmarketable, but mine waste materials often contain environmentally significant concentrations of heavy metals and precious metals.

This report describes the major potential impacts on the quality of public drinking water sources associated with the various elements of mining. It is recognized that some discussion may not accurately reflect the environmental conditions at modern hardrock mining operations that are well designed, operated, and regulated. The intent of the discussion is to describe environmental problems that may occur at historic, current, and future mine sites.

Mining Methods

Precious metals and industrial metals typically occur in disseminated ore bodies or vein deposits. The two primary methods used to mine metals and minerals include surface or open-pit mining and underground mining. Surface or open-pit mining is typically used for large shallow ore bodies, which have a low metal or mineral value per volume of rock. Underground mining is typically used when the mineralized rock is deep and occurs in veins.

Surface or open-pit mining often requires the removal and disposal of soil and rock overburden that contains no target mineral. The underlying ore body typically includes some rock that contains uneconomical concentrations of the target mineral. This waste rock is also removed and typically

[1] Regional Ground-Water Expert, U.S. Environmental Protection Agency, Region VIII, Denver, CO.

stockpiled or otherwise disposed of. The portion of the ore body to be mined is drilled, blasted, and transported to a facility where it is crushed and prepared for milling or leaching.

Underground mining requires the excavation of vertical shafts, horizontal adits, and inclined adits to access the ore body. The rock that is excavated during adit construction is commonly referred to as development rock. Once the ore body is reached, horizontal passages called drifts and crosscuts are developed on numerous levels and the ore is mined. Waste rock and ore are transported to the surface via rails or small trucks, or they are hoisted to the surface in vertical shafts.

In both surface and underground mining, extraction of ore waste materials requires heavy equipment and explosives. A commonly used explosive is a mixture of ammonium nitrate and fuel oil. As the development rock, waste rock, and ore are removed, they are typically transferred to large trucks for transportation to storage or processing facilities. Overburden and development rock usually do not contain minerals that reduce the quality of surface or ground water, and they can be used as mine backfill, but they are typically disposed of in piles near the mine site. Waste rock from the ore body can contain environmentally significant amounts of metals and should be tested for acid-generating potential. It is important to segregate waste rock that is potentially acid generating and not use it for mine backfill or impoundment dams.

Surface and underground mines typically extend below the local and regional water table or both. As a result, ground water may flow into the mine pit or underground workings. Water collecting in the mine pit or workings must be removed. In open-pit mines, this water is typically pumped out and discharged to nearby surface water. In underground mines, the water can be pumped out or drainage adits can be constructed at or below the lowest mine level to allow for free drainage of the water entering the workings. Many precious metal ore bodies occur in mountainous terrain where the host rock is commonly comprised of igneous or metamorphic rocks. In these types of rocks, ground water occurrence and flow is controlled by the distribution and orientation of fractures, joints, and faults. In these settings, ground water inflow into mine workings occurs only where the mine workings intersect water-bearing structures.

Ore Processing

Ore processing, or milling, refers to the processing of ore rock to create the size of the desired product, remove unwanted constituents, and concentrate or otherwise improve the quality of the desired product. Applicable milling processes are determined based on the physical and chemical properties of the target metal or mineral, the ore grade, and environmental considerations.

Amalgamation

This is the process where metallic mercury is added to gold ore to separate the gold from the ore rock. When liquid mercury comes in contact with gold, it bonds with the surface of the gold particles (amalgamation). The mercury-coated gold particles coalesce or collect into a gray, plastic mass. When this mass is heated, the mercury is driven off and the metallic gold remains.

Flotation

The physical and chemical properties of many minerals allow for separation and concentration by flotation. Finely crushed ore rock is added to water containing selected reagents. These reagents create a froth, which selectively floats some minerals while others sink. Common reagents include copper, zinc, chromium, cyanide, nitrate, phenolic compounds, and, for copper ore, sulfuric acid. The waste (tailings) and the wastewater are typically disposed of in large, constructed impoundments.

Leaching

Leaching refers to processes that involve spraying, pouring, or injecting an acid or cyanide solution over crushed and uncrushed ore to dissolve metals for later extraction. The type of solution used depends on the ore's physical and chemical characteristics. Leaching is used almost exclusively on low-grade ore. The main types of leaching include dump, heap, and in situ leaching. For each type, a nearby holding area (typically a pond) is used to store the pregnant solution prior to recovery of the desired metal by a chemical or electrical process. Once the desired metal is recovered, the solution is reused in the leaching process.

In dump leaching, the material is generally piled on the ground, and the leaching solution is applied to the pile by spraying, injecting, or washing. Dump leach piles can be very large, often covering hundreds of acres (hectares) and containing millions of tons of ore rock. Leaching solutions aided by precipitation dissolve the desired metals. Dump leach piles are not placed on clay or synthetic liners. The pregnant solution drains away from the bottom of the leach pile to a holding pond. Pregnant solution can be lost to the subsurface, which reduces the amount transported to the holding pond, and potentially contaminates ground water. Dump leaching is used for very low-grade ore.

Heap leaching is used for higher grade ores and is generally conducted on a smaller scale than dump leaching. The ore is usually crushed and is placed on a pad constructed of synthetic materials or clay. These low-permeability liners help maximize recovery of the leachate.

In situ leaching involves pumping a reagent (commonly a sulfuric acid solution) directly into the ore body. The reagent dissolves the desired mineral, and the pregnant solution is collected and pumped to the surface for extraction of the desired mineral.

Leaching can recover economic quantities of the desired mineral for months, years, or decades. When leaching no longer produces economical quantities of metals, the spent ore is typically rinsed to dilute or otherwise detoxify the reagent solution to meet environmental standards. If standards are met, the rinsing may be discontinued and the leached material may be allowed to drain. The spent ore is then typically left in place.

Water Management

Management of water at large mine sites is a critical element of a mining operation. At large mine sites that include a mill and a tailings impoundment, water management is difficult. It is complicated by the many management requirements, which may include the dewatering of open pits and underground mine workings or both, the transportation of surface runoff across mine sites, the use and containment of water used for ore processing, and the need to meet applicable water-quality standards for all discharges from the mine site. Historically, the management of water has not focused enough on prevention of environmental impacts. Nationwide, there have been numerous incidents where contaminated water from a mine site has been improperly discharged, impairing the quality of surface water.

Waste Management

Hardrock mining typically produces large volumes of solid waste, including overburden, development rock, waste rock, spent ore, and tailings. Overburden, development rock, and waste rock are typically stockpiled at the mine site. Some of these materials may be used as pit backfill or uncommonly for backfill of underground workings. Overburden and development rock usually pose minimal threats to the environment. Waste rock can contain significant concentrations of metals and pyrite and may present an environmental problem. Some waste rock stockpiles may be left in place for future ore processing.

Tailings are the waste solids remaining after ore processing. Tailings generally leave the mill as slurry consisting of 40 to 70 percent liquid and 30 to 60 percent fine-grained solids. Tailings can contain significant concentrations of heavy metals and other contaminants. Most tailings are disposed of in on-site impoundments. Historically, tailing impoundments were not lined and were located without consideration of potential environmental impacts. Modern tailing impoundment design often includes low-permeability clay or synthetic liners, engineered caps designed to eliminate or minimize infiltration of water into the tailings, and collection systems to capture leachate that escapes from the impoundment.

Seepage from tailing impoundments is often unavoidable and raises the probability of surface water and ground water contamination. Such seepage and acid rock drainage may require water treatment long after the active life of the facility. Failure to maintain adequate hydrostatic pressure within and behind an impoundment dam may result in failure of the impoundment structure, releasing tailings and effluent to surface and ground water.

Spent ore is a waste material that is generated at mines that utilize heap or dump leaching. The volume of spent ore can be very large and can contain environmentally significant residual amounts of leaching reagent and dissolved metals. Both spent ore and tailings need to be actively managed for years after mine closure to ensure that leachate does not escape to a nearby stream or infiltrate into underlying ground water.

Mine Closure

Closure of a mining operation occurs during temporary shutdown of operations or permanent decommissioning of the facilities. Depending on the type of mine, the size and nature of the area of disturbance, and the type of ore processing, active management of the mine site including water management may be necessary for years or even decades following closure. Until recently, reclamation was limited to grading and revegetating waste materials and pits to minimize erosion and improve the visual landscape. Permanent closure now routinely includes some or all of the following: removal and disposal of stored fuels and chemicals, structure tear down, removal of unnecessary roadways and ditches, shaft and adit plugging, waste detoxification, capping of tailings, backfilling pits, and active water management to ensure that all applicable water-quality standards are met. In numerous cases, a water treatment facility must be operated and maintained. At mine sites where acid mine drainage is a problem, water treatment may be necessary for decades.

The long-term nature of mining impacts may require that environmental monitoring (source, early warning, and compliance monitoring), contingency planning, and financial insurance be in place for decades. Geochemical conditions in the ore body, waste rock, tailings, and workings can change over time. Hence, the ability is needed to make necessary changes in water control and water treatment after mine closure.

Issues and Risks

At hardrock mines, adits and shafts, underground workings, open pits, overburden, development rock and waste rock dumps, tailings impoundments, leach pads, process ponds, and mills are known sources of heavy metals, sulfate, cyanide, and nitrate. If released in environmentally harmful concentrations, these contaminants can have significant negative effects on the quality of surface water and ground water for public drinking water sources. Dissolved and total metals concentrations can impact public water supplies and the aquatic health of stream and riparian systems.

Surface runoff is a key mechanism for release of pollutants into streams and lakes. Seepage from tailings ponds and waste rock piles, unwanted releases from process water ponds or wastewater ponds, drainage from underground workings, and discharge of pit water may contaminate water resources. Surface waters may also be impacted by contaminated ground water or contaminated by heavy metals in sediments. The mobility of contaminants is increased by exposure to rain and snowmelt.

A variety of complex geochemical and hydrogeological processes control the transport, attenuation, and ultimate distribution of heavy metals and other mine-related contaminants. Dissolved and suspended contaminants are transported to aquifers and streams via complex overland and subsurface pathways. This complexity, combined with the large scale of mining activities and the numerous mine-related sources of contaminants, make water-quality assessments and restoration and remediation of mine sites very difficult.

Environmental problems are often more difficult to deal with at abandoned mine sites that lack environmental monitoring. Several thousand abandoned and inactive mines exist on public land. The U.S. Department of Agriculture, Office of Inspector General estimates that there are more than 38,000 abandoned and inactive hardrock mines on land administered by the Forest Service.

The major types of water-quality impacts include erosion and sedimentation, acid rock drainage, cyanide leaching, and dissolution and transport of toxic metals. These impacts are discussed in the following sections.

Erosion and Sedimentation

Because mining may disturb large areas and expose large quantities of earthen materials, erosion and subsequent transport of sediment to surface water can be a major concern. Major sources of erosion and sedimentation include open-pit areas, heap and dump leach piles, overburden, development and waste rock piles, tailings piles and dams, haul and access roads, ore stockpiles, vehicle and equipment maintenance areas, exploration areas, and reclamation areas. Historically, erosion and sedimentation have built up thick layers of mineral fines and sediment in floodplains and streams at many mine sites. These sediments can carry attached chemical pollutants and toxic metals, which can be stored in floodplain and bed sediments. To avoid these problems, erosion and sedimentation must be controlled from the beginning of operations through postclosure treatments.

Sediments and minerals deposited in floodplains can impact the quality of nearby surface water and underlying ground water. Oxidation of sulfide minerals may lower the pH of surface runoff, thereby mobilizing heavy metals that can infiltrate into underlying ground water and/or be transported to nearby surface water. Reduced soil pH also may kill riparian vegetation.

Drinking water impacts associated with erosion and sedimentation are discussed in chapter 2.

Mining disturbances also can increase surface runoff, which can result in increased streamflow velocities and volumes, downstream flooding, scouring of stream channels and structural damage to water diversions, drinking water intakes, bridge footings, and culverts.

Acid Rock Drainage

A major water-quality problem at hardrock mine sites is the formation of acid rock drainage and the associated mobilization of toxic metals, iron, sulfate, and total dissolved solids. The formation of acid rock drainage results from the exposure of sulfide minerals (pyrite, pyrrhotite, galena, sphalerite, and chalcopyrite) to air and water. Sulfide minerals are commonly associated with coal deposits and precious and heavy metal ore bodies. Pyrite (FeS), the most common sulfide mineral, reacts with water and oxygen to produce ferrous iron (Fe^{+2}), sulfate (SO_4), and acid (H^+). In waters where oxidizing conditions are prevalent and the pH is >3.5, ferrous iron will oxidize to ferric iron. Much of the ferric iron precipitates as iron hydroxide. Some ferric iron remains in solution and continues to chemically accelerate the oxidation of pyrite and subsequent generation of acid. As the pH continues to decrease, the oxidation of ferrous iron decreases and the precipitation of iron hydroxide decreases. This results in a greater dissolved concentration of ferric iron and, therefore, a greater rate of sulfide (pyrite) oxidation. The oxidation of sulfide minerals is also catalyzed by Thiobacillus ferrooxidans bacteria. These bacteria, which are common in the subsurface, can increase the rate of sulfide oxidation by 5 or 6 orders of magnitude. When low pH water comes in contact with metal-bearing rocks and minerals, a number of toxic metals dissolve and are transported by the water. Different metals are dissolved over different ranges of pH. The most common metals associated with sulfide minerals include lead, zinc, copper, cadmium, and arsenic.

Both water and oxygen are necessary to generate acid drainage. Water is both a reactant and a medium for the bacteria that catalyze the oxidation process. Water also transports the oxidation reaction products and the associated dissolved metals. Atmospheric oxygen is a very strong oxidizing agent and is important for bacterially catalyzed oxidation at pH values below 3.5. Surface water and shallow ground water typically have relatively high concentrations of dissolved oxygen.

Acid rock drainage can be discharged from underground mine workings, open-pit walls and floors, tailings impoundments, waste rock piles, and spent ore from leaching operations. Acid rock drainage occurs at both active and abandoned mines. Acid generation and drainage of acid water with high concentrations of dissolved metals affect both surface and ground water. Ingesting water contaminated by heavy metals can have significant health affects for humans and aquatic organisms, including water birds and fish. Metals and other mine-related contaminants in sources of drinking water can exceed water-quality standards. Expensive treatment or acquisition of another source of water may be the only alternatives.

Cyanide Leaching

For over a century, cyanide has been used as a pyrite suppressant in base metal flotation and in gold extraction. Dump leaching and heap leaching operations commonly use cyanide in the leaching solution. Continued improvements in cyanide leaching technology have allowed the economic mining of lower grade ores. As a result, increasing amounts of cyanide are being used in mining. The mining industry now uses most of the sodium cyanide used in the United States. More than 100 million pounds (45 million kilograms) were used by gold and silver leaching operations in 1990.

Cyanide can cause two major types of environmental impacts: (1) ponds and ditches (and to a lesser degree, tailings impoundments) that contain process water containing cyanide solutions can present an acute hazard to wildlife, especially aquatic birds; and (2) spills or other unwanted releases of cyanide solution from ponds, leach impoundments, spent ore piles, or tailings impoundments can enter surface water killing fish and contaminating drinking water sources. During the 1980’s and early 1990’s as the use of cyanide leaching increased worldwide, a number of serious cyanide spills and unwanted releases have occurred. Impacts on wildlife and streamwater quality have been significant. These incidents and the acute toxicity of cyanide have focused public attention on the use of cyanide in the mining industry.

When cyanide is inhaled or ingested, it interferes with an organism’s oxygen metabolism and can be lethal in a short time. Cyanide is much more toxic to aquatic organisms than to humans. The acute aquatic standard is 22 milligrams (mg) per liter and the chronic aquatic standard is 5.2 mg per liter. The maximum contaminant level for public drinking water supplies is 200 mg per liter. These values are for total cyanide even though toxicity is caused by free cyanide. Total cyanide is usually measured because it is difficult to measure free cyanide. Nitrate, a breakdown product of cyanide, is also a drinking water problem (see chapter 2).

Cyanide that is dissolved in water readily complexes with metals. At pH values below 9, weaker cyanide compounds can dissociate and hydrogen cyanide (HCN), a volatile poisonous gas, is formed as a byproduct. If cyanide-contaminated water infiltrates into unsaturated soil and the pH of the water is lowered to below 9, free cyanide can volatilize to hydrogen cyanide. Cyanide can also be attenuated to some degree by other processes, including adsorption, precipitation, oxidation to cyanate, and biodegradation.

Once the leaching of ore dumps or heaps is complete, it is necessary to rinse the spent ore until the appropriate cyanide standard is reached. In arid regions, getting enough water to rinse heaps or dumps can be a significant problem. In wet climates, excess water from heavy precipitation can increase the risk of unwanted cyanide releases from leach dumps or heaps. The chemistry of the spent ore and the associated water in leaching impoundments can change over time, creating a potential for continued release and transport of dissolved metals long after the cyanide concentration has been reduced by rinsing. Factors affecting the chemistry of a heap leaching impoundment include pH, moisture, and ore mineralogy.

Also of significant concern is the long-term structural stability of large heap leach impoundments. The physical characteristics of the leached ore, the physical configuration of the impoundment, and specific site conditions affect the long-term structural stability of a leach impoundment. Structurally unstable impoundments may fail, allowing contaminated leachate or sediments to reach public drinking water sources.

Transport of Dissolved Contaminants

Dissolved contaminants (primarily metals, sulfate, and nitrate) can migrate from mining operations to underlying ground water or nearby surface water that is a source for drinking water. Discharges of process water, mine water, runoff, and seepage from mine waste piles or impoundments can transport dissolved contaminants to source water.

Under specific conditions, dissolved constituents in surface water can precipitate and attach to sediments. Elevated concentrations of lead and mercury are often found in sediments while being undetected in the water column. Sediment contamination may affect human health through consumption of fish that bioaccumulate toxic pollutants. Contaminated sediment provides a long-term potential source of pollutants that, under certain geochemical conditions, can dissolve in the water column.

The likelihood of contaminants dissolving and migrating from mine waste materials or mine workings to ground water depends on the nature and management of the waste materials, the local hydrogeologic setting, and the geochemical conditions in the underlying vadose zone and aquifer. Risks to human health and the environment from contaminated ground water can be significant. In many hydrogeologic settings, ground water discharge provides a significant percentage of stream baseflow. In this manner, ground water contaminated by mining activities can also contaminate surface water.

At some locations, naturally occurring substances in an ore body can be a significant source of contaminants. The rocks that comprise ore bodies contain varying concentrations of nontarget minerals, including radioactive minerals. Other minerals may be present at concentrations that can be toxic and can be mobilized by the same geochemical and hydrological processes that control transport of mine-related contaminants. Nontarget minerals that can pose a risk to drinking water sources include aluminum, arsenic, asbestos, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, silver, selenium, thallium, and zinc. Unlike many other types of industrial operations and associated discharges, contaminant loading from hardrock mine sites can vary significantly with the season.

Findings from Studies

During the past 10 years, an increasing number of environmental studies have characterized the environmental impacts associated with active, inactive, and abandoned hardrock mines. Most of these studies have focused on water-quality impacts. In 1995, the Bureau of Land Management, the U.S. Geological Survey (USGS), and the Forest Service jointly developed a strategy to address cleanup of abandoned mines on Federal land (Nimick and von Guerard 1998). As part of this strategy, the USGS developed an abandoned mine land initiative that included numerous pilot studies in the Boulder River watershed in Montana and the Animas River watershed in Colorado. Most of the applied research efforts associated with this initiative were aimed at determining sources and magnitudes of metal loadings in nearby streams. A number of these studies documented significant metal loading from mining-related facilities and also from unmined areas underlain by sulfide ore bodies.

Using authorities under the Clean Water Act and Comprehensive Environmental Response, Compensation and Liability Act (CERLA), the U.S. Environment Protection Agency (EPA) has conducted a number of studies in Colorado and Montana. These studies have characterized the hydrologic pathways and geochemical processes that control the release and transport of toxic metals from mining facilities to underlying ground water and nearby surface water. Allen and Stanley (1998) summarize water-quality data collected in 1974–97 from streams that flow out of the New World Mining District in southwestern Montana. Water quality in two different streams has been significantly impacted by metals loading from mine workings, mine waste, and, to a lesser degree, by “natural” background loading (metals mobilized and transported to streams in the absence of any mine-related disturbance). In Daisy Creek, which flows past a mine pit that has been backfilled with mine waste, dissolved copper concentrations have ranged from 0.93 to 6.22 mg per liter at a location just downstream from the mine pit. The average concentration in 13 samples was 2.24 mg per liter. The drinking water standard is 1.0 mg per liter, and the chronic aquatic standard is 0.012 mg per liter. Dissolved iron concentrations at the same location ranged from 0.55 to 12.30 mg per liter. The average concentration in 13 samples was 3.53 mg per liter. The drinking water standard for iron is 0.30 mg per liter.

Studies conducted by EPA's Region VIII and the Colorado Division of Minerals and Geology in the Chalk Creek mining district in southern Colorado have documented extensive metal loading to Chalk Creek from the historic Mary Murphy gold mine. Zinc loading attributed to the extensive underground workings in Chrysolite Mountain and mine waste piles in the floodplain of Chalk Creek has been well documented. Zinc concentrations as high as 192,300 micrograms (µg) per liter have been measured in leachate from an old tailings pile less than one-fourth of a mile from Chalk Creek.[2] Data from 1999 indicate excessive zinc levels at three locations: (1) as high as 32,730 µg per liter in water discharging from the portal of the Golf Tunnel, which is the lowermost adit in Chrysolite Mountain; (2) 221,300 µg per liter in ground water seeping down through the upper workings in Chrysolite Mountain; and (3) 341 µg per liter in Chalk Creek below the Mary Murphy mine.[3] The drinking water standard for zinc is 5,000 µg per liter and the chronic aquatic standard is 110 µg per liter (at 100 mg-per-liter hardness). It is clear from these data that mining activities have had a significant impact to ground water and surface water in the vicinity of the Mary Murphy mine.

Reliability and Limitations of Findings

Data and information on potential environmental impacts related to hardrock mining have increased greatly in the past 10 years. Numerous investigations and published reports have documented movement of toxic metals to ground water and surface water from mines and mine-related facilities. The data from the increasing number of reports is reliable because the findings are comparable and often present the same conclusions. Many of the study results have been published in peer-reviewed literature.

One point of disagreement and uncertainty is the significance of "natural" background metal loadings versus metal loadings that result from mining activities. A number of studies have attempted to separate "natural" from man-caused loading (Nimick and von Guerard, 1998). Researchers have used water-quality data, including isotopes and tracers, to try to identify loading caused by leaching of unmined ore bodies. However, to date there has been no reliable technique developed to clearly separate natural from man-caused loading.

Research Needs

1. Research needs related to the environmental management of hardrock mine sites include two primary areas: (a) characterization of hydrologic and geochemical processes that control the release and transport of mine-related contaminants away from a mine site to ground water or nearby surface water; and (b) development of workable, passive systems for treating water with low pH and high concentrations of dissolved metals.
2. Hardrock mines often occur in complex hydrogeologic settings where a standard approach to characterization of ground water and surface water is inadequate. A mine can greatly disturb natural hydrologic systems, creating major water pollution problems. It is critical that we continue to improve characterization approaches and tools. An increased understanding of processes, which control distribution of mine-related contaminants, will be helpful for planning future mines and implementing effective environmental controls.
3. Capital, operating, and maintenance costs associated with active treatment of contaminated mine water are prohibitive at most mine sites. It is extremely important to continue research directed at developing efficient and cost-effective passive treatment technologies that can be operated year-round at high elevations. Research must continue on the use of organic substrata to facilitate the utilization of sulfide-reducing bacteria to remove dissolved metals from water. To date these technologies have been limited by the inability to deal with high-flow rates and the extreme climatic conditions at high elevations.

---

[2] Science Applications International Corporation. 1993. Chalk Creek nonpoint source project case history. 99 p. Unpublished report prepared for U.S. Environmental Protection Agency, Region VIII, Denver, CO. On file with: Science Applications International Corporation, 999 18th Street, Denver, CO 80202–2405.

[3] Wireman, Mike. 1999. Unpublished field data from Mary Murphy mine—Chalk Creek Mine District, Chaffee County, CO. [Not paged]. On file with: Mike Wireman, U.S. Environmental Protection Agency, Region VIII, 999 18th Street, Suite 500, Denver, CO 80202–2405.

Key Points

1. Management practices are commonly used to control erosion and sedimentation at mine sites. The selection of erosion control measures is based on site-specific considerations, such as facility size, climate, geographic location, geology, hydrology, and the environmental setting of each mine site. Mining facilities are often in remote locations and may operate only seasonally or intermittently, but they need year-round pollution controls. At least six categories of management practices are available to limit erosion and the off-site transport of sediment including discharge diversions, drainage and stormwater conveyance systems, runoff dispersion, sediment control and collection, vegetation and soil stabilization, and capping sources of contamination.
2. No easy or inexpensive solutions to acid rock drainage are currently available. An appropriate approach is to isolate or otherwise segregate waste with acid-generation potential, and then treat them appropriately. Management may include minimizing contact with oxygen and water and/or neutralizing acid that is produced with natural or introduced material. Techniques used include subaqueous disposal, covers, waste blending, hydrologic controls, bacterial control, and treatment.
3. Acid-generation prediction tests are increasingly relied upon to assess the long-term potential of pit walls and floors, underground workings, and mine waste to generate acid. Mineralogy and other factors affecting the potential for acid rock drainage are highly variable from site to site, and this can result in less than accurate predictions. In general, the methods used to predict the acid-generation potential are classified as either static or kinetic. Static tests are intended only to predict the potential to produce acid rather than predict the rate of acid generation. Static tests can be conducted quickly and are inexpensive compared with kinetic tests. Kinetic tests are intended to mimic the processes found in the environment of the ore body or waste unit environment; however, they require more time and are more expensive than static tests. Reliable dynamic tests that are faster and less expensive are needed.
4. The heightened awareness of the potential environmental problems associated with cyanide leaching led Federal land managers and States to implement increasingly stringent regulations and guidelines. These regulations and guidelines address the design of facilities that use cyanide and include requiring or recommending use of liners with heap leach piles or tailings impoundments, monitoring of solutions in process waters and ponds, treatment requirements for cyanide-containing wastes, and closure and reclamation requirements. Operators are generally required to take steps either to reduce or eliminate unwanted releases of cyanide solutions or to reduce cyanide concentrations in exposed materials to below standards. Regulatory requirements and guidelines on the allowable concentration of cyanide in exposed process solutions vary. When numeric limitations are established, they generally range around 50 mg per liter.

Literature Cited

Allen, Bryan K.; Stanley, Dan R. 1998. Site assessment and sampling activities report—New World Mine, Cooke City, Montana. URS Operating Services. Contract 68–W5–0031. Washington, DC: U.S. Environmental Protection Agency. 108 p.

Lyon, J.S.; Hilliard, T.J.; Bethell, T.N. 1993. Burden of gilt. Washington, DC: Mineral Policy Center. 24 p.

Nimick, David A.; von Guerard, Paul, eds. 1998. Science for watershed decisions on abandoned mine lands; 1998 February 4–5; Denver. U.S. Geo. Surv. Open File Rep. 98–297. Washington, DC: U.S. Government Printing Office. 71 p.

Chapter 19

Coal Mining

Mike Wireman[1]

Introduction

The mining of coal can have many of the same environmental impacts to water quality as hardrock mining. However, some aspects of coal mining are different enough to warrant a separate discussion. After a brief description of coal mining, this chapter focuses on aspects of coal mining that are significantly different from hardrock mining with regard to the potential to impact water quality.

Coal accounts for one-third of the total energy usage and more than one-half of the electricity generated in the country (U.S. Geological Survey 1996). Domestic coal production has been steadily increasing since the 1950's. In 1998, total domestic production was 1.18 billion tons.[2] Approximately 570.5 million tons were produced in States east of the Mississippi River and 547.6 million tons from States west of the Mississippi River. Coal production in the West has almost doubled since the passage of the 1991 Amendments to the Clean Air Act. Wyoming leads the Nation in coal production. West Virginia and Kentucky are second and third, respectively. About 60 percent of domestic production is from surface mines and 40 percent from underground mines.

Mining Methods

Strip mining is the most common method of producing coal from surface mines. Strip mining commonly includes the removal and storage of topsoil, the removal of any overburden material, and the subsequent excavation of the coal seam. As the operation advances across the land surface, only a relatively small area is actively mined. With this method, the overburden is removed from the advanced side of the active mine face and placed on the retreat side, where the coal has been mined out. There are two common methods of underground mining: room and pillar mining and longwall mining. In the room and pillar method, entries or adits are driven into the coal seam, and crosscuts are driven at right angles to the adits at spacings dictated by the individual mine plan. A checkerboard pattern of interconnected tunnels or rooms and pillars is created. In longwall mining, numerous crosscuts are developed around a large block of coal. Once the crosscuts are fully developed, the large block is completely excavated, and the chamber is allowed to collapse. Longwall mining results in predictable subsidence of the overlying ground surface.

Coal Preparation

Coal that is excavated from a seam or deposit requires preparation to improve the quality and make it suitable for a given use. Preparation includes the separation of the heavier waste material from the lighter coal by flotation processes that rely on the differential densities of the coal and the waste material. Reagents are sometimes used to make the coal more amenable to flotation. Coal preparation creates a relatively uniform product size, reduces the amount of ash in the coal, and may reduce the sulfur content. In addition to clean coal, the preparation process produces a coarse, dewatered waste rock material and a fine-grained slurry with significant water content.

Waste Management

Waste materials are generated from coal mining and coal preparation. Overburden material removed for surface mining is often used to backfill the excavated area. Waste materials from underground mining are disposed of in mined-out workings to the extent possible, but they often are placed in a designated waste rock disposal area on the surface.

Large volumes of waste material can be generated from coal preparation. Both the coarse waste rock and the fine-grained slurry are typically disposed of in disturbed portions of the permit area. The fine slurry waste is commonly disposed of in an impoundment where the slurry solid settles, and the water is reclaimed from pond on top of the impoundment.

[1] Regional Ground-Water Expert, U.S. Environmental Protection Agency, Region VIII, Denver, CO.
[2] Personal communication. 1999. Stuart Sanderson, President, Colorado Mining Association, 216 16th Street, Suite 1250, Denver, CO 80202.

Environmental Regulation

With the passage of the Surface Mining and Control Reclamation Act (SMRCA) in 1977, the coal mining industry became the only mining sector in the United States that is subject to mine-specific environmental regulation. The SMCRA pertains only to coal and was promulgated by the U.S. Congress to provide environmental standards for reclaiming land that has been impacted by coal mining and processing operations. The Office of Surface Mining Reclamation and Enforcement (OSM) was established to administer the law and regulations established by SMCRA. The OSM can delegate the regulatory program to the State level and most States that have substantial coal resources have developed their own regulatory programs. Most States have developed a permit program that regulates exploration activities, surface mining, underground mining, and special mining activities.

Issues and Risks

Just as in precious metal mining, the mining of coal can result in the exposure of sulfide minerals to oxygen, water, and bacteria. Pyrite and less commonly marcasite (FeS_2) and greigite (Fe_3S_4) are the primary sulfide minerals found in coal. Oxidation of these minerals can result in the generation of acidic water and the subsequent mobilization and transport of heavy metals to ground water and surface water.

Mine waste and coal preparation waste can contain significant amounts of pyrite and heavy metals including cadmium, chromium, mercury, nickel, lead, and zinc. These metals and sulfur can be concentrated in waste materials by factors of 3 to 10 compared to raw coal (National Research Council Committee on Accessory Elements 1979). Therefore, just as in hardrock mining acid drainage, the associated mobilization of heavy metals in the waste materials is a potentially significant threat to surface and ground water resources. See chapter 18 for further discussion of acid drainage and heavy metal mobilization.

Findings from Studies

The scientific literature includes thousands of studies on water-quality impacts from the mining and processing of coal. Coal mining has been much more extensively studied in the United States than hardrock mining. In the Southeastern United States where coal mining has occurred for more than 100 years, there are numerous documented cases of contamination of streams from coal mining. Hyman and Watzlaf (1997) used water-quality data from 128 different samples of untreated coal mine drainage from mines in Pennsylvania, West Virginia, Ohio, Tennessee, Maryland, Montana, Kentucky, Colorado, Oklahoma, and Missouri to characterize the occurrence of various metals and other contaminants. Results from this study indicate that the mean concentrations for arsenic, beryllium, cadmium, and lead exceeded the maximum contaminant level for drinking water and the maximum concentrations of these metals plus antimony, chromium, and zinc exceeded the maximum concentration level. This study also concluded that the traditional use of manganese concentrations as an indicator parameter for treatment thresholds is not reliable and that water-quality protection is better achieved if individual metal concentrations are more thoroughly considered.

Reliability and Limitations of Findings

It is clear that coal mining can mobilize and transport toxic metals from mines and mine-related facilities to ground water and surface water.

Research Needs

1. Within the coal mining industry, a key focus of recent environmental research has been the environmental effects of surface mining and power generation. A significant amount of research has involved mining and reclamation because these activities have the greatest impact on the environment. Major environmental concerns faced by the coal industry include the impacts of surface mining on water resources and whether mined land can be returned to productive use for crops, livestock, timber, and wildlife (White and others 1997). Important areas of research include topsoil substitution, reforestation, forage and row crops production, and wetlands. All of these areas of research are aimed at providing a better understanding of how areas that have been disturbed by coal mining can be reclaimed to reduce impacts on water quality.
2. An area of research that needs to be expanded is the development of methods for characterizing the hydrologic and geochemical processes that control release and transport of mine-related contaminants away from a mine site to ground water or nearby surface water. This research need is similar to that for hardrock mining. More emphasis needs to be given to preventing or controlling the transport of contaminants to streams.

Key Points

1. Mining and processing of coal clearly have the potential to contaminate ground water and nearby surface water. The mobilization and transport of toxic metals and other contaminants has been well documented in many areas of the country, especially in the leading coal-producing States in the Southeast. In Kentucky, West Virginia, Tennessee, and Virginia, the potential to impact surface water quality is increased by steep topography and narrow valleys. In this terrain, it is very difficult to mine and process coal without impacting surface water.
2. In the Western United States, coal production has increased significantly since 1991. In general, the western coal has a low sulfur content, which reduces the potential for acid rock drainage. In addition, the geologic and topographic settings of coal deposits in Western States is generally more amenable to the implementation of environmental controls.
3. The SMCRA requires all coal operations to develop environmental information, file operation and reclamation plans, and post an adequate surety prior to the development of any coal mining operation. Management practices are commonly used to control erosion and sediment at mine sites. Traditionally, the focus of the reclamation has been to restore the land disturbed by coal mining to beneficial use. Since the passage of the Clean Water Act, coal mining operations have been subject to point-source permitting. However, as with hardrock mining, no easy or inexpensive solutions to controlling acid rock drainage are currently available. Isolating materials will help to prevent or minimize oxygen contact with the material and prevent water from contacting the material.

Literature Cited

Hyman, David M.; Watzlaf, George R. 1997. Metals and other components of coal mine drainage as related to aquatic life standards. In: Brandt, J.E., ed. Proceedings of the American Society for Surface Mining and Reclamation 14th annual national meeting; 1997 May; Austin, TX. Austin, TX: American Society for Surface Mining and Reclamation: 531–545.

National Research Council Committee on Accessory Elements. 1979. Redistribution of accessory elements in mining and mineral processing. 1: Coal and oil shale. Washington, DC: National Academy of Science. 180 p.

U.S. Geological Survey. 1996. U.S. Geological Survey programs in Montana. USGS Fact Sheet FS–026–96. Washington, DC: U.S. Government Printing Office. 4 p.

White, R.L.; Smith, S.F.; Miller, P.J. 1997. TU system environmental research program. In: Brandt, J. E., ed. Proceedings of the American Society for Surface Mining and Reclamation 14th annual national meeting; 1997 May; Austin, TX. Austin, TX: American Society for Surface Mining and Reclamation: 680–684.

Chapter 20

Oil and Gas Development

R.J. Gauthier-Warinner[1]

Introduction

Oil and gas exploration generally has short-term effects on the quality of drinking water sources. Exploration consists of geologic mapping and ground geophysical methods consisting of surface gravity, magnetic, and seismic surveys of the prospective area. Gravity and magnetic data are obtained with little impact to the surface. Seismic surveys entail the stringing of numerous arrays of geophones and the drilling of relatively few shot holes for creating the seismic signals. Today, the seismic energy often is generated by thumpers mounted on large trucks, utilizing less-environmentally damaging vibroseis technology. Both methods require a system of crude roads for access; however, vibroseis does not require the logistical support or involve the site disturbance that is necessary for drilling.

The Bureau of Land Management (BLM) oversees drilling operations and specifies conditions that must be met during drilling on public land. These conditions are designed to meet the intent of specific laws, such as the Safe Water Drinking Act of 1996, as well as to mitigate negative effects on resources that may not be specifically protected under statute or regulation.

Exploratory well drilling entails both site occupancy and reconfiguration. It has relatively short-term effects. Exploratory wells can acquire drill cuttings and cores for visual analysis as they probe the formation for direct information about such rock characteristics as lithology, porosity, permeability, and identification of pore fluids.

The majority of well drilling in today's petroleum industry is accomplished with rotary drills. This type of drilling requires the circulation of a fluid to lubricate and cool the bit, prevent plugging of the hole, and maintain the necessary hydrostatic pressure to prevent collapse of the well. It also counterbalances any high-pressure oil, gas, or water encountered in any of the drilled formations. Thus, fluid circulation helps prevent a catastrophic surge of highly pressurized fluid, called a blowout. Blowouts can cause fires, loss of life and property, and potential contamination of surface drinking water sources.

The fluid circulation system uses drilling muds. Generally, they are a water-based mixture of clays like bentonite and inert weighting constituents like barite with special additives mixed in low concentrations. Formulation of a particular drilling mud is based upon downhole conditions such as drilling depth, temperature, pressure, and the sensitivity of an oil or gas reservoir to water. Weighting constituents are added to the mud to counterbalance the formation pressure and prevent the formation fluids from entering the wellbore. The drilling mud is circulated downward through the drill stem, into the bit, and back up the annular space between the drill stem and the hole. It is then screened, filtered, and recirculated through tanks back into the hole. The Forest Service has some discretion in requiring that certain conditions be met in fluid system design and location. There is a risk of contamination to an intervening freshwater aquifer. The magnitude of risk depends, among other things, on the competence of the oil- and gas-containing rock, the proximity of the aquifer, and the thickness and competence of the units separating them.

Lined earthen pits, unlined earthen pits, or closed circulation systems are used for containment of water, waste fluids from drilling, rock cuttings, rigwash, and stormwater runoff. Containment design is influenced by such factors as soil conditions, depth to freshwater aquifers, proximity to surface water sources and drainages, types of drilling fluid, and availability of water for drilling. The design, location, closure, and reclamation of containment systems for drilling operations fall under the jurisdiction of both the Forest Service and the BLM.

Upon reaching the desired depth, the well is analyzed by electric and nuclear logs to determine whether the hole is a potential producer. If the well is determined to have no potential for production, it is plugged. Plugging operations fall under the jurisdiction of the BLM and the State or both. The responsible agency must ensure that plugging meets local criteria for protection of underground water sources. If the well is determined to have potential for production, the production casing is cemented into the wellbore and the

[1] Geology Program Manager, USDA Forest Service, Minerals and Geology Staff, Washington, DC.

drilling rig is replaced with a smaller completion rig. Casing a wellbore serves several purposes. It prevents the formation from caving into the wellbore; it provides a permanent passageway for conveying the oil and gas to the surface; it prevents exotic fluids from mixing with the producing formation; and it isolates the producing zone or other contaminating zones in the well from contact with any freshwater aquifers penetrated by the well. Casing operations fall under jurisdiction of the BLM and the State or both. They must be in compliance with specifications designed to protect underground water sources and to contain high pressures and any fluids or gases or both that might escape to the surface and pose hazards to surface resources, including drinking water sources.

Once drilled, cased, and completed, many wells have insufficient force to flow without further assistance because of material introduced by drilling or of material within the formation itself. Two of the most common techniques of well stimulation are acidizing and fracture treating. Acidizing is the pumping of acid into the well to help dissolve the impediment. When the permeability of a reservoir is so low that it is difficult for the oil and gas to flow into the well, the rock may be fractured to allow oil and gas to flow freely to the wellbore. A high-pressure fracture fluid comprised of thickened or gelled water is pumped at high rates into the well to fracture the formation.

After a well is completed for production, the drill is removed from the site and replaced by the well head. This phase of the operation has long-term effects because the facilities associated with it are in place over the operating life of the well. The Forest Service takes on long-term responsibilities for administering ongoing operations and monitoring conditions under which the operations occur.

Equipment design and layout are tailored to the particular characteristics of the site and the type of production (oil, gas, oil/gas mixtures; associated water production; oil/gas components such as hydrogen sulfide; etc.). The emphasis is on containment of fluids and gases, particularly in emergency circumstances. Although specific types of equipment are continuously being designed or upgraded to provide for environmentally safe production operations, it is often not practical, economical, or necessary to retrofit existing operations with some of the newer technology. The Forest Service must work closely with the BLM and the State or both in developing conditions of approval under which production facilities can be safely constructed and operated.

A flowing well is any well that has sufficient pressure belowground to cause the oil or gas to flow unassisted through the wellbore to the surface. Artificial lift is a technique that employs a mechanical or artificial means to pump or lift the oil to the surface. Depending upon the particular circumstances associated with the well, one of several types of artificial lift can be used. Primary recovery is the initial production of fluids using only natural sources of energy available within the reservoir. Depending upon the natural reservoir energy available, primary recovery can range from <5 percent to 75 percent of the resource. Secondary and tertiary recovery includes utilization of such methods as injection of water, steam, carbon dioxide, polymers, or micellar fluids to supplement natural reservoir energy and increase fluid recovery.

Generally, oil produced from the well is a mixture of oil, water, gas, and sand or other solid material. The sand and other solid materials are generally removed by gravity methods. Typically, the oil and water occur as an emulsion and must be treated to break the emulsion. Several methods are used for this purpose. Heaters can be used to heat the emulsion and separate it into its oil and water constituents. The addition of certain types of chemicals or the use of direct current can facilitate this process.

Once at the surface, the product is transferred by gathering lines to be treated then stored in underground or surface tanks until it is shipped to the purchaser. Storage facilities are comprised of welded or bolted steel tanks of various sizes ranging from 50 barrels to more than 10,000 barrels, depending on the scale of production. Facilities typically include provisions for transfer to trucks or pipelines. Refer to the discussion of roads and utility corridors in chapter 9.

Gas reservoirs generally do not contain oil, but produce gas with varying amounts of condensate or water. They generally produce well without the addition of supplementary energy and primary recovery methods are usually sufficient. Recovery is often >80 percent of the resource.

Issues and Risks

The Forest Service has a limited role in administering oil and gas operations. It has surface responsibilities only; whereas, the BLM, the U.S. Environmental Protection Agency (EPA), and the State have jurisdiction over subsurface operations. Additionally, the Forest Service can only make recommendations to the BLM regarding whether or not to issue a lease and what stipulations to apply if leased. The BLM has no obligation to implement Forest Service recommendations. The Forest Service must work closely with the BLM and the State in developing conditions of approval under which production facilities can be constructed and operations can be maintained.

The Resource Conservation and Recovery Act of 1976 (RCRA), codified at 42 U.S.C. sec. 6901 et seq., conditionally exempted from regulation as hazardous wastes drilling fluids, produced waters and other wastes associated with the exploration, development, or production of crude oil or natural gas. According to the EPA, exempted wastes include well completion, treatment, and stimulation fluids; workover wastes; packing fluids; and constituents removed from produced water before it is injected or otherwise disposed of. While these wastes are not considered hazardous, they may have an effect on the quality of drinking water sources if contamination occurs. Contamination is most likely to occur at the surface in the event of a spill or a breach of, or infiltration from, a containment structure.

Access roads and well pads erode and become sources of sediment during the exploration and production phases. See chapter 9 for discussion of roads and sediment.

Under the Clean Water Act, discharges to surface water by oil and gas exploration and production operations are addressed by the National Pollutant Discharge Elimination System. Onshore discharges are prohibited except from wells producing not more than 10 barrels per day and discharges of produced water that are determined to be beneficial to agriculture or wildlife (U.S. EPA 1992).

The Safe Drinking Water Act specifically addresses oil and gas operations under its underground injection control program. The objective of the program is to protect good-quality ground water from contamination by injected fluids. It established a special class (class II) of injection wells for oilfield-related fluids, the regulation of which should not impede oil and gas production unless necessary to prevent contamination of underground sources of drinking water. An underground source is an aquifer that supplies drinking water for human consumption or for any public water system, or contains fewer than 10,000 milligrams per liter of total dissolved solids, does not contain minerals or hydrocarbons that are commercially producible, and is situated at a depth or location, which makes the recovery of water for drinking purposes economically or technologically practical. Class II regulatory programs are either directly administered by the States under primacy programs or by EPA where States do not administer the programs.

Injection wells are sometimes used to dispose of produced water, a byproduct of oil and gas recovery. Most produced water is strongly saline, with total dissolved solids ranging from several hundred to over 150,000 parts per million (ppm). Produced water pumped into injection wells is used to enhance production by providing the energy needed to drive the oil toward the producing well. Secondary recovery may necessitate the drilling of a few to hundreds of injection wells throughout the field, depending upon the size of the reservoir. This water is intended to provide the energy needed to drive the oil toward the producing well. Secondary recovery may necessitate the drilling of a few to hundreds of injection wells throughout the field, depending upon the size of the reservoir.

Because produced water is beneficially recycled and is an integral part of some crude oil and natural gas production processes and because injection of produced water for enhanced recovery is regulated under the Safe Drinking Water Act's Underground Injection Control Program, EPA has determined that it is not a waste for purposes of RCRA subtitle C or subtitle D.

Despite prevention measures, contamination of a drinking water aquifer can occur as a result of improper plugging of abandoned wells or casings, and through direct injection into aquifers. During exploratory and development drilling, the well has the potential to act as a conduit between formations hosting usable aquifers and formations containing hydrocarbons, heavy metals, or chlorides associated with accompanying brines. If the well penetrates an aquifer and is not cased, or the casing and grouting fail, there is a possibility for contaminants to migrate through the conduit and into the drinking water aquifer.

Stimulation of an oil reservoir utilizing the pumping of a fracture fluid under high pressure into the formation can have adverse effects. If the induced fracturing extends beyond the boundaries of the reservoir, there is a risk of contamination to a nearby freshwater aquifer. The magnitude of risk is dependent, among other things, on the competence of the reservoir rock, proximity of the aquifer, and the thickness and competence of the units separating them.

Produced water is usually a highly saline brine accompanied by trace contaminants inherent in the reservoir. Injection of produced water back into the reservoir for disposal or to enhance recovery has the potential to contaminate freshwater through grout or casing failures between the injection well and the aquifer. Injecting produced water into old injection wells with leaking casings can introduce brine into surface geologic strata where it can percolate to and contaminate surface waters. Sometimes brine water is trucked to injection wells; however, some truckers have been known to dump the brine illegally into surface water at stream crossings.

Corrosion or failure of any one of the numerous surface facilities may result in leakage and subsequent migration of hydrocarbons into shallow freshwater aquifers. Surface pipes from wells to storage tanks can corrode or break and discharge oil and brine onto the soil surface, where the discharge can run off to streams. Pipes crossing streams can rupture and discharge directly into streams. The degree of contamination depends upon, among other things, the extent and duration of leakage.

Some waste management practices associated with hydrocarbon production may have an effect on ground water. The failure of waste pits or drilling mud pits or the utilization of unlined pits for these purposes can allow percolation of contaminants through the soil and into shallow aquifers. Some natural gas contains hydrogen sulfide, carbon dioxide, or other impurities that must be removed prior to sale. Sweetening is the stripping of these impurities by various chemical processes including utilization of amine, sulfinol, iron sponge, and caustic solutions. Associated wastes may include spent amine, gycol and sulfinol, slurries of sulfur and sodium salts, iron sulfide and wood shavings, and caustic filter material, which may be commingled with produced water. These wastes may fall into a hazardous waste category but are exempted from regulation under RCRA. Any waste products associated with oil and gas production, whether exempted or not, can be a risk to drinking water sources if not managed appropriately.

The disposal of excess drilling fluid and produced water by evaporation, road spreading, and application to the land may have an effect on the quality of surface water. Runoff may allow the migration of chlorides, oily wastes, or other contaminants into streams or ground water and, thus, affect the quality of drinking water.

Findings from Studies

With respect to the disposal by landspreading of liquid and solid wastes, two primary concerns are their salt content and hydrocarbon content. Studies by Deuel (1990) and Macyk and others (1990) have shown that soil and water mixtures or both with soluble salt levels below roughly 3,000 ppm of total dissolved solids, exchangeable sodium percentage of < 16, and a sodium adsorption ratio of < 12 cause no harm to soil, vegetation, surface water, or ground water. Landspreading or wastes resulting in oil and grease concentrations of up to 1 percent by weight in the waste and soil mixture or both are not harmful and will biodegrade readily. Repetitive disking and nutrient addition can reduce concentrations in a soil mixture to these levels.

Instream monitoring by the Daniel Boone National Forest in Kentucky revealed high concentrations of brine below oil production well fields. In Texas, heavy sediment deposits in streams were traced to gas well pads and service roads.

Reliability and Limitation of Findings

Anecdotal evidence of contamination or degradation of drinking water sources from oil or gas wells exists throughout the Forest Service, particularly in areas of split mineral estates in which the Federal Government holds surface rights, but mineral rights are privately owned. Such estates are most common in the national grasslands and eastern national forests. Contamination or degradation has not been assessed on a nationwide scale, but the level of risk depends on the degree of monitoring and inspection. Databases managed by the BLM and individual States may provide more information about the extent of existing contamination or degradation and potential for such to occur in the future.

Research Need

A quantified assessment of contamination or degradation of surface and ground water by oil and gas operations that draws on BLM and State data bases is needed on a nationwide scale. It would provide a more accurate framework in which to manage oil and gas exploration and production activities.

Key Points

All facets of oil and gas exploration and production can affect the quality of drinking water. The Forest Service can control the effects associated with those activities that occur on the land surface such as site preparation, berm and pit construction, design and location of ancillary systems, road construction, and reclamation activities that probably have a greater potential to affect surface water quality. The Forest Service must work closely with the BLM, EPA, and the States to assure that drilling, production, and waste disposal activities are conducted so as to minimize adverse effects on both ground water and surface water quality for public drinking water sources.

Literature Cited

American Petroleum Institute. 1993. Environmental guidance document: well abandonment and inactive well practices for U.S. exploration and production operations. 1st ed. Washington, DC: American Petroleum Institute. 52 p.

American Petroleum Institute. 1996. Introduction to oil and gas production. 5th ed. Washington, DC: American Petroleum Institute. 120 p.

American Petroleum Institute. 1997a. Environmental guidance document: onshore solid waste management in exploration and production operations. 2d ed. Washington, DC: American Petroleum Institute. 84 p.

American Petroleum Institute. 1997b. Exploration and production: protecting the environment. Washington, DC: American Petroleum Institute. 24 p.

Deuel, L.E. 1990. Evaluation of limiting constituents suggested for land disposal of exploration and production wastes. In: Proceedings of the 1st international symposium on oil and gas exploration waste management practices; 1990 September 10–13; New Orleans. Washington, DC: American Petroleum Institute: 411–430.

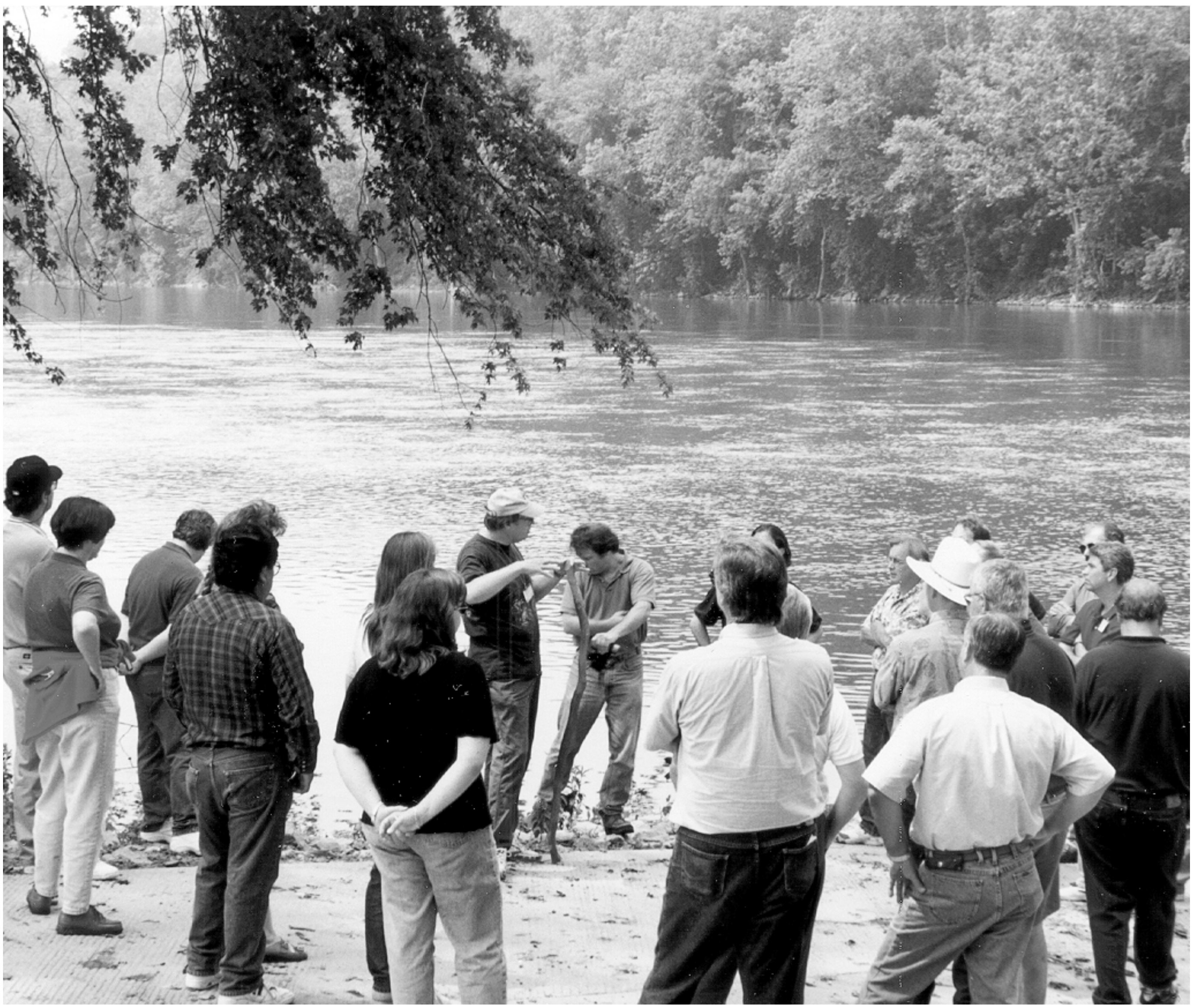
Macyk, T.M.; Nikiforuk, F.I.; Weiss, D.K. 1990. Drilling waste landspreading field trial in the Cold Lake Heavy Oil Region, Alberta, Canada. In: Proceedings of the 1st international symposium on oil and gas exploration waste management practices; 1990 September 10–13; New Orleans. Washington, DC: American Petroleum Institute: 267–279.

Interstate Oil and Gas Compact Commission. 1997. Groundwater risk assessment and remediation. Oklahoma City: Interstate Oil and Gas Compact Commission. [Number of pages unknown].

U.S. EPA. 1992. Background for NEPA reviewers: crude oil and natural gas exploration, development, and production. EPA Contract 68–WO–0025. Falls Church, VA: Science Applications International Corporation, Environmental Health Sciences Group. 85 p.

Part VI:

Implications for Source Water Assessments and for Land Management and Policy



A U.S. Fish and Wildlife Service class on "working at the watershed level" at the Potomac River near Sheperdstown, WV. Photo by Stephen Glasser

Chapter 21

Future Trends and Research Needs in Managing Forests and Grasslands as Drinking Water Sources

F.N. Scatena[1]

Introduction

The management of forest and grassland watersheds for drinking water supplies has been, and will continue to be, a major activity of the Forest Service and other natural resource agencies. However, these watersheds will continue to support other uses, including providing timber products, recreation, mining, fisheries, grazing, and the conservation of biodiversity. In addition, relatively new uses like using forests for carbon and nutrient sequestration (DeLucia and others 1999) or the recycling of wastewater (Cole and others 1986, Sopper and Kardos 1973) will increase. The future is also expected to bring increased competition for existing water resources (Postel 1998) and changes from point source to watershed-based pollution management (U.S. EPA 1997). How these watersheds will be managed in this increasingly competitive, watershed-based, multiuse environment will be affected by site-specific knowledge of environmental change, technological change, and social and administrative considerations.

Environmental Change

It is widely believed that the Earth is undergoing a period of rapid global climate change that will significantly alter environmental conditions in many areas during the 21st century (Schlesinger 1997). Most global-scale climate models predict that in the next 50 years the Earth will be warmer, more humid, and have greater evaporation, precipitation, and runoff (Loaiciga and others 1995). However, not all areas will be affected equally, and large areas of the United States may actually experience more arid conditions. Projections based on historic hydrologic conditions and projected demands suggest that areas east of the Great Plains and in the Pacific Northwest will have water surpluses until 2040 (Guldin 1989). In contrast, much of the Colorado and the Rio Grande River basins, the Great Basin, parts of California, and the lower Mississippi Valley currently have or will have water shortages by 2040. Moreover, 11 out of 18 water resource regions of the Continental United States are currently diverting more than 20 percent of their streamflow for off-site uses and will be affected by changes in either streamflow or water demand (U.S. Department of Agriculture, Forest Service 2000). The availability of water is predicted to seriously constrain global food production by 2025 (Postel 1998). Water shortages and aquifer depletion already affect many of the World's most important food-producing regions, including the Western United States, northern China, the Punjab of India, and parts of Southeast Asia, Africa, and the Middle East.

In addition to global-scale change, local and regional environmental change can be expected to influence municipal water supplies. These changes can occur over years or decades and may include changes in land uses or increases in air pollution and atmospheric inputs. Increases in upstream water withdrawals or changes in the types of water uses (irrigation, snow making, etc.) can also modify water quality and temporal patterns of streamflow. Changes in the successional status of forest cover or the presence of exotic or noxious weeds can also affect source water quantity and quality (see chapter 11). Shifts in management practices can also influence local environmental conditions that affect municipal water supplies. Increased harvesting on steeper slopes, growing wood in high-input, short-rotation plantations, and intensively managing fisheries are some examples (see chapters 10, 17).

In response to the influence of local and regional environmental changes, assessments are now being developed to evaluate regional risks of specific environmental hazards (Graham and others 1991). A generic problem encountered when doing these assessments is the lack of ecosystem-specific information on the spatial and temporal variability of ecological and pollution-generating processes. Fortunately, technology advances in data acquisition and in the management of spatially explicit data using Geographical Information Systems (GIS) is rapidly improving this situation. Nevertheless, an administrative commitment to long-term environmental monitoring, data analysis, and

[1] Ecosystem Team Leader, USDA Forest Service, International Institute of Tropical Forestry, Río Piedras, PR.

synthesis is required to develop adequate assessments and verify ecological and resource management models.

Technological Change

The ability to provide safe drinking water depends on how technology is used to: (1) measure the quantity and quality of water, (2) run treatment plants and distribution systems, and (3) charge consumers for the water they use and the pollution they generate. Advances in the technology used to accomplish these tasks are expected to improve municipal water management. Meanwhile, other anthropogenic activities can be expected to create new chemicals, new pathogens, and presently unknown water-quality problems.

Increased competition for water resources and increased emphases on instream water quality are anticipated to increase the scrutiny of water supply and pollution management. Fortunately, recent advances in the technology used to acquire hydrologic data are greatly improving our ability to measure and monitor water resources. At the regional scale, advances in climate modeling and remote sensing have increased the ability to monitor precipitation and water resources over large areas. At the local scale, automated sensors and wireless communication systems are monitoring streamflows and water withdrawals, nutrient and pollution concentrations, stream channel morphology, and the migration of aquatic organisms. Moreover, real-time monitoring is currently being used to manage irrigation systems, water supply reservoirs, flood warning systems, and biotic migrations in rivers. Future developments are expected in the technology to monitor pesticides, special chemicals, and microbiological constituents. If pollution standards based on total maximum daily watershed loads replace standards based on average point-source discharges (U.S. EPA 1991), we would expect improvements in the technology for low-cost monitoring of temporal variations in water quality.

New technology and scientific understanding can be combined to improve the timing of land-use treatments. Recent examples of this type of management include: (1) the timing of streamwater withdrawal and release to minimize impacts on the migration of aquatic organisms (Benstead and others 1999, Bistal and Ruff 1996); (2) the timing of fertilizer application to growth phases of agricultural crops to minimize nutrient runoff (Matson and others 1998); (3) timing the abundance of grazing with the growing season of range and riparian vegetation (chapter 14); and (4) the scheduling of insecticide application to the life cycles of pests (Balogh and Walker 1992; chapter 13). The success of these life-history-based management schemes requires detailed knowledge of local environmental conditions; high-quality, spatially explicit monitoring; and institutional memory of past successes and failures.

While automated data collection techniques are essential to accurately monitoring hydrological and ecological processes, it will remain technically and economically impossible to monitor and treat for all contaminates, at all locations, at all times. Furthermore, without proper analysis, the automated collection of massive amounts of data can hinder rather than assist management. A major challenge for watershed management in the 21st century will be the development of spatially explicit analytical methods and institutional structures that can rapidly synthesize information about environmental conditions so managers can make informed, defensible decisions in a timely fashion. In response to this challenge, many natural resource organizations are developing data management and environmental decision-support systems (Lovejoy and others 1997, Spencer 1996). These decision-support systems will eventually integrate real-time hydrologic measurements with GIS and multiobjective decision models. Multiobjective models have been used for decades in the design and operation of reservoirs and water and wastewater distribution systems because they provide a formal and logical structure for organizing and synthesizing scientific, environmental, and social information (Hipel 1992). Nevertheless, their adoption as to real-time management tools will be a considerable challenge. Developing and verifying site-specific models to establish maximum daily pollutant load allocations for specific management practices or individual landowners will also be a major research and management challenge (U.S. EPA 1991, 1997). Moreover, just collecting real-time, water-quality information to be used in complex models can be a daunting task (Bistal and Ruff 1996).

Administrative Change

Because of high engineering and environmental costs associated with developing new water supplies, the emphasis in water management is shifting from the development of new sources toward the efficient and equitable use of existing supplies (Frederick 1993, Kneese 1993). Likewise, the costs and risks associated with transforming polluted water into potable water are increasing the emphasis on maintaining source water quality. In response to these shifts, the administrative structures and organizational relationships used to manage municipal water systems are also changing. One reflection of this change is the number of regional water management councils, stormwater utilities, waste management districts, and watershed restoration groups that have recently been established (Mann 1993, Shabman 1993, Taff and Senjem 1996) (see examples in appendices A, B). These new organizations typically develop to help manage the complexities associated with the mismatch between natural water regions, political districts, and the geography of water demand and wastewater generation.

While new organizational structures are developing in some regions, many traditional water and pollution management organizations are caught in a vicious cycle (World Bank 1993). Because of unreliable and poor-quality services, consumers are unwilling to support increase tariffs for water management and pollution control. Inadequate operating funds lead to further deterioration of services by the overextended agencies. To successfully deal with these problems, economists have championed "user-pays" and "market-based" approaches to water and pollution management for over 45 years (Busby 1955, Kneese 1993). For municipal water systems, this approach has typically meant establishing variable rate structures to promote conservation and/or the privatization of water management services. Several different market-based approaches have been promoted for pollution control, including the use of effluent charges, tradable effluent permits, tax on the use of substances that threaten water quality, and open competition for interbasin or interregion water or pollutant transfers (Mann 1993, Taff and Senjem 1996).

While these market-based approaches have been widely promoted by economists, water resource managers have been less enthusiastic about their adoption (Brookshire and Neill 1992, Taff and Senjem 1996). The principal stumbling blocks are usually the technological and organizational requisites for monitoring and enforcing the complex and dynamic trade of water resources and pollutant effluent. Recently developed data acquisition technology allows pollution discharges and water withdrawals to be monitored continuously. It, therefore, is generally believed that these market-based, watershed-based approaches will increase in the near future as water-quality regulations based on watershed-wide maximum pollution loads are implemented.

Site-Specific Considerations

The social and environmental responses to changes that can affect municipal watersheds are complex because different ecosystems and processes respond at different rates and in different magnitudes (Schimel and others 1996). Furthermore, the risk and consequences of inaccurate decisions are not evenly allocated across the landscape or population (Frederick 1993). Therefore, site-specific research and monitoring are needed to develop local and ecosystem-specific understandings of the processes that effect water quality at a particular location**.** For example, removal of a road requires site-specific analysis, or the disturbance caused by road closure may accelerate rather than reduce erosion (Elliot and others 1996) (see chapter 9). Likewise, the impacts of grazing also require an understanding of the specific grazers and the local, seasonal cycle of rangeland vegetation (see chapter 14). The response of atmospheric deposition of nitrogen also depends on the level of nitrogen saturation of the receiving ecosystem and seasonal variations in plant growth and nitrogen use (Fenn and others 1998). Toxic algal blooms in lakes can occur in specific portions of some lakes and result from unique combinations of site-specific climatic events and management operations (James and Havens 1996) (see chapter 5, Hebgen Lake case study). The abundance and frequency of herbicide use also depend on site conditions and can range from never to several times each year (Balogh and Walker 1992) (see chapter 13). Furthermore, the width and composition of buffer zones needed to contain certain chemicals, reduce impacts of grazers, or maintain aquatic habitat also are ecosystem and problem specific. Water-quality changes due to ozone-induced stress on conifers are also closely related to location-specific forest cover changes (Graham and others 1991). Likewise, the impacts of accidental chemical spills or other historical legacies that alter water quality are site-specific and require local knowledge and institutional memory to be properly assessed and efficiently managed.

The importance of understanding the timing of specific environmental events and processes is an additional theme in many of the chapters in this report. For example, the amounts and impacts of recreation on site and water quality are regional, seasonal, and episodic (see chapters 6, 7, and 8). The use and, therefore, the potential influence of wildlife on water quality can also vary with seasonal and diurnal behavior and the abundance of specific populations (see chapters 15, 16, and 17). Grazing behavior also varies with season, species, and the age of individuals (chapter 14). Likewise, determining life histories and vectors of waterborne human pathogens is also essential for evaluating site-specific risks and management options.

The length of time that a particular activity affects water quality also varies with land use and site-specific characteristics. Sediment yields or concentrations following timber harvesting typically decrease as a negative exponential relationship while changes in nutrient concentrations occur in relatively brief pulses (see chapter 10). Sediment yields from roads typically peak in the first few years but can remain elevated for decades, while contamination from roadside fuel spills can last for years (see chapters 9, 11). Likewise, the residence time of fecal contamination in streams can be on the order of weeks to months (see chapter 2). Mining debris can acidify surface and subsurface water for decades or longer (see chapters 18, 19).

Developing methods and monitoring protocols to determine and predict environmentally critical time periods is an additional challenge that will require the interaction of scientific information, technology, and administrative structures. The future success of these interactions will depend on the availability of high-quality, spatially explicit, long-term environmental data. In many regions, assessments of municipal water supplies will provide invaluable baseline information for future managers.

Conclusion

The flow of water across the landscape and its collection and distribution through a municipal water system are complex and dynamic processes. Because of the complexities and risks involved, some basic level of water treatment and monitoring is always necessary. However, it is technically and economically impossible to monitor and treat for all contaminates, at all locations, at all times. Providing the necessary levels of watershed protection, treatment, and monitoring to sustain supplies has been, and will continue to be, a major challenge. The dramatic improvements in U.S. water quality that have occurred during the last few decades clearly demonstrate the success that integrated, continued management can have. It is hoped that this report will assist water resource managers to successfully identify critical problems and protect the best and restore the rest.

Literature Cited

Balogh, J.C.; Walker, W.J. 1992. Golf course management and construction. Environmental issues. Chelsea, MN: Lewis Publishers. 951 p.

Benstead, P.J.; March, J.G.; Pringle, C.M.; Scatena, F.N. 1999. Effects of a low-head dam and water abstraction on migratory tropical stream biota. Ecological Applications. 9(2): 656–668.

Bistal, G.A.; Ruff, J.D. 1996. Quality of 1995 spring total dissolved gas data: Columbia and Snake Rivers. Water Resources Bulletin. 32(6): 1177–1186.

Brookshire, D.S.; Neill, H.R. 1992. Benefit transfers: conceptual and empirical issues. Water Resources Research. 28(3): 651–655.

Busby, C.E. 1955. Regulation and economic expansion. In: Water: the 1955 yearbook of agriculture. Washington, DC: U.S. Department of Agriculture: 666–676.

Cole, D.W.; Henry, C.L.; Nutter, W.L., eds. 1986. The forest alternative for treatment and utilization of municipal and industrial wastes. Seattle: University of Washington Press. 356 p.

DeLucia, E.H.; Hamilton, J.G.; Naidu, S.L. [and others]. 1999. Net primary production of a forest ecosystem with experimental CO_2 enrichment. Science. 284: 1177–1179.

Elliot, W.J.; Foltz, C.H.; Luce, C.H.; Koler, T.E. 1996. Computer-aided risk analysis in road decommissioning. In: McDonnell, J.J.; Stribling, J.B.; Neville, L.R.; Leopold, D.J., eds. Proceedings of the AWRA annual symposium on watershed restoration management: physical, chemical, and biological considerations; 1996 July 14–17; Syracuse, NY. Herndon, VA: American Water Resources Association: 341–350.

Fenn, M.E.; Poth, M.A.; Aber, J.D. [and others]. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecological Applications. 8(3): 706–733.

Frederick, K.D. 1993. The economics of risk in water resource planning. In: Reuss, M., ed. Water resources administration in the United States: policy, practice, and emerging issues. East Lansing, MI: American Water Resources Association, Michigan State University Press: 182–194.

Graham, R.L.; Hunsaker, C.T.; O'Neill, R.V.; Jackson, B.L. 1991. Ecological risk assessment at the regional scale. Ecological Applications. 1(2): 196–206.

Guldin, R.W. 1989. An analysis of the water situation in the United States: 1989–2040. Gen. Tech. Rep. RM–177. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest Experiment Station. 178 p.

Hipel, K.W. 1992. Multiple objective decision making in water resources. American Water Resources Association Monograph. 18: 1–231.

James, T.R.; Havens, K.E. 1996. Algal bloom probability in a large subtropical lake. Water Resources Bulletin. 32(5): 995–1006.

Kneese, A.V. 1993. Economics and water resources. In: Reuss, M., ed. Water resources administration in the United States: policy practice and emerging issues. East Lansing, MI: American Water Resources Association, Michigan State University Press: 23–35.

Loaiciga, H.A.; Valdes, J.B.; Vogel, R. [and others]. 1995. Global warming and the hydrological cycle. Journal of Hydrology. 174: 82–127.

Lovejoy, S.B.; Lee, J.G.; Randhir, T.O.; Engel, B.A. 1997. Research needs for water quality management in the 21st century. Journal of Soil and Water Conservation. 52(1): 18–21.

Mann, D.E. 1993. Political science: the past and future of water resources policy and management. In: Reuss, M., ed. Water resources administration in the United States: policy, practice, and emerging issues. East Lansing, MI: American Water Resources Association, Michigan State University Press: 55–65.

Matson, P.A.; Naylor, R.; Ortiz-Monasterio, I. 1998. Integration of environmental, agronomic, and economic aspects of fertilizer management. Science. 280: 112–115.

Postel, S.L. 1998. Water for food production: will there be enough in 2025? Bioscience. 48(8): 629–637.

Schimel, D.S.; Braswell, B.H.; McKeown, R. [and others]. 1996. Climate and nitrogen controls on the geography and timescales of terrestrial biogeochemical cycling. Global Biogeochemical Cycles. 10(4): 677–692.

Schlesinger, W.H. 1997. Biogeochemistry: an analysis of global change. 2d ed. San Diego: Academic Press. 588 p.

Shabman, L. 1993. Bargaining, markets, and watershed restoration: some elements of a new national water policy. In: Reuss, M., ed. Water resources administration in the United States: policy, practice, and emerging issues. East Lansing, MI: American Water Resources Association, Michigan State University Press. 314 p.

Sopper, W.E.; Kardos, L.T., eds. 1973. Recycling treated municipal wastewater and sludge through forest and cropland. University Park, PA: Pennsylvania State University Press. 479 p.

Spencer, T.A. 1996. Information management: a framework for the future. In: Neary, D.; Ross, K.C.; Coleman, S.S., eds. National hydrology workshop proceedings; 1992 April 27–May 1; Phoenix, AZ. Gen. Tech. Rep. RM–279. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest Experiment Station: 205–207.

Taff, S.J.; Senjem, N. 1996. Increasing regulator confidence in point-nonpoint pollutant trading schemes. Water Resources Bulletin. 32(6): 1187–1193.

U.S. Department of Agriculture, Forest Service. 2000. Water and the Forest Service. FS–660. [Washington, DC]: [U.S. Government Printing Office]. 26 p.

U.S. EPA. 1991. Guidance for water quality-based decision: the TMDL process. EPA–440–.4–91–001. Washington, DC: U.S. Environmental Protection Agency. 116 p.

U.S. EPA. 1997. State source water assessment and protection programs guidance. EPA–816–R–97–009. Washington, DC: U.S. Environmental Protection Agency. 128 p.

World Bank. 1993. Water resources management. A World Bank policy paper. Washington, DC: The World Bank. 140 p.

Chapter 22

Synthesis

Douglas F. Ryan[1]

Introduction

Forest and grassland watersheds have traditionally been relied upon as sources of drinking water with relatively little contamination. Recent results from the National Water-Quality Assessment (U.S. Geological Survey 1999) reaffirm that, nationwide, water from forest and grassland watersheds is lower in many pollutants than that from watersheds dominated by urban or agricultural land use. These findings do not contradict the scientific evidence reviewed in this report, which shows that many types of common land-use practices and natural processes in forests and grasslands can introduce contaminants into water sources. Rather, taken together, these results indicate that although land-use practices in forests and grasslands can introduce contaminants, the characteristics and intensity of these practices, when applied over large areas, produce water that is cleaner in many respects than other, more intensive land-use practices. At the local level, forest and grassland management may cause significant problems for drinking water sources. For example, high-intensity activities such as logging, mining, or urban-style development in forests can cause considerable pollution as can uncontrolled events such as floods, landslides, or accidental chemical spills. At the regional level, contaminants from forests and grasslands, even where low in concentration, are part of the overall, cumulative load of water pollution. Thus, assessing the risk of contamination for drinking water systems with source areas in forests or grasslands is not fundamentally different from assessing risks in areas with other types of land uses. Regardless of land use, assessments should be done on a case-by-case basis, analyzing the natural processes and human activities that can reasonably occur in the source area to estimate the likelihood that contaminants will be transmitted to a drinking water intake.

Drinking Water Contaminants and Treatments

Contaminants of concern for drinking water have been classified and standards for acceptable levels set by the U.S. Environmental Protection Agency (EPA). The relationship between specific forest and grassland best management practices and drinking water quality is complex and was not treated in detail in this report. Best management practices and the protection that they provide for water quality vary considerably from State to State and are evolving over time. As effects on human health from 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. Standards for drinking water do not apply to source water before it has been treated to remove contaminants. Standards set under the Clean Water Act (Public Law 80–845) that apply to ambient water as it flows in a stream or lake are not intended to ensure that water is drinkable without treatment. Considerable treatment may be required to purify water that meets the ambient standard to make it comply with the drinking water standard. For examples of forest and grassland management practices that have been proposed to protect water quality, see U.S. EPA (1993), copies of which can be ordered from the Web at http://www.epa.gov/OWOW/info/PubList/publist4.html.

Drinking water treatment technology can be designed to reduce most contaminants in source water to an acceptable level before delivery to consumers. The cost of treatment, however, usually increases substantially as the amount of contamination in source water increases. Adopting appropriate land uses and management practices that do not contaminate source water has the potential to be more cost-effective than treatment of source water that has been contaminated.

Cumulative Effects

Although different types of land use are treated separately in this report, in an individual watershed many different land uses affect source water quality simultaneously. Land uses occur in complex patterns that overlap on the landscape and change over time. Relatively few studies have examined the

[1] Staff Watershed Specialist; Wildlife, Fish, Watershed and Air Research Staff; USDA Forest Service; Washington, DC.

cumulative effects of several land uses distributed over time and space. Most studies have focused on relatively small areas and short time periods. Some of the tools that are needed to analyze cumulative effects at large scales have only become available recently. More information is needed on the interactions among multiple land uses in complex and changing patterns because complexity is typical of watersheds in which land management is likely to affect public drinking water sources.

Effects of Natural Processes and Human Activities

Drinking water sources are affected by numerous natural and human-influenced processes that occur in watersheds. The processes by which water and contaminants move through watersheds are relatively well known. The risks these processes pose to source water can be severe, especially during extreme natural events such as floods and landslides.

Water quality may be affected where water is dammed, impounded, diverted, or augmented for a variety of human purposes that may or may not be related to drinking water use. These activities often alter flow rates and residence time of water. They change turbidity, sediment storage and transport, and oxygen content of water. Considerable information is available about effects of these manipulations on source water, but this information must be applied on a case-by-case basis. Removal of old dams may pose risks for drinking water sources downstream. Sediments and toxic contaminants or both in the material accumulated behind the dam can be mobilized. Few studies, however, have evaluated this risk.

Intermixed urban and wildland uses, developed administrative sites, and concentrated recreational sites share a number of similar risks for source water. Runoff from impervious surfaces and improperly functioning sewerage treatment facilities can contaminate surface water, while poorly performing septic systems and leaking underground fuel storage tanks can pose risks for both surface and ground water. Systems that are old, inadequately designed, and/or poorly maintained represent the highest risks. Forms of concentrated recreation that involve direct water contact, such as bathing beaches, may have a high potential for contaminating surface water but have been little studied.

Dispersed recreation activities that attract people to spend time near water bodies, but lack developed sanitary facilities, can introduce fecal organisms, presumably including pathogens, into surface waters. However, few studies have been done to determine thresholds above which dispersed use contaminate source water excessively. Many other potential effects of dispersed recreation on drinking water sources, such as risks associated with pets and off-road vehicles, are poorly understood and need further research.

Roads and other utility corridors pose risks of contamination because they concentrate many human activities. Roadside recreation facilities are often centers of dispersed recreation. Roads also support transport of chemicals, some of them toxic. Chemicals may be spilled during accidents; the highest risk of water contamination is where roads cross streams. Utility corridors present similar risks due to incidents such as pipeline failures or transformer fires that may spill chemicals. Road and corridor construction, maintenance, and use have been shown to contribute sediments to streams because they have elevated erosion rates and can increase the risk of landslides on unstable terrain. Proper engineering design, construction and maintenance of roads and utility corridors, as well as emergency preparedness can reduce but not entirely eliminate these risks to source waters.

Many researchers have studied the impacts on water quality from manipulating forest vegetation for purposes such as timber and fiber production including growing trees, harvesting them, and reestablishing forest vegetation. The primary contaminant to source water from these activities is sediment associated with soil disturbance during harvesting and regeneration and erosion from roads. Enrichment of streamwater by nitrate after forest harvesting has been reported in some parts of the country. Mounting evidence suggests that this response in some regions may be explained by mobilization of long-term accumulations in forest soils of nitrogen compounds that were deposited from air pollution. The degree and areal extent of this effect needs further study.

Most modern pesticides and herbicides that are currently used on forests and grasslands are immobilized and degraded in soils to the extent that they pose little contamination risk to source water if required application precautions are followed. Even though the use of pesticides that resist breakdown in the environment has largely been discontinued, some of these substances may persist in the landscape as a result of past use. They may pose problems for source water, especially if deposits of these chemicals in soils or sediments are mobilized by disturbance.

Prescribed fire is normally conducted under conditions when fire severity is low and impact on source water quality under these conditions also is low. Wildfires, however, can be severe. When they occur on steep or erodible terrain and are followed by intense rainfall, they can produce large sediment loads that pose problems for source waters. The causes of high nitrate levels in streams after fires in areas with high air pollution, and the effects of fire retardant chemicals on source water need further study. The effectiveness of emergency rehabilitation practices to stabilize watersheds after severe wildfires also needs more investigation.

Land management that results in domestic or wild animals being concentrated near surface water can contaminate source water, but in most cases the risk of transmission to source water is little understood. Domestic grazing animals, such as cattle, have been shown to introduce sediment and fecal organisms into surface waters. Streamside buffers and other practices that reduce contact between grazing animals and surface water can reduce contamination risks. Although many wildlife species are known to carry pathogens, relatively little is known about the risk that wildlife-carried pathogens pose for source water. The few studies of wildlife effects on water quality mainly involve ungulates, such as elk and deer that pose risks analogous to domestic grazing animals when wild ungulates reach high concentrations near surface waters. Fish hatcheries can introduce fecal matter and chemicals if these substances are flushed from hatchery facilities to source waters. Less-intensive forms of fisheries management, such as altering fish habitat by introducing large woody debris and restoring habitat for anadromous fish runs, may also have effects on source water but these aspects have been little studied. Large concentrations of water birds can introduce pathogens into surface water, and measures are sometimes taken to discourage water bird congregations near drinking water intakes.

Mining has the potential to contaminate source water with sediments, acids, toxic metals, and other introduced chemicals from mining residues and ore processing. Many of these effects may be abated or mitigated during active mining but can pose long-term risks to source water if mines are not properly decommissioned after mining ceases. Oil and gas exploration and extraction can cause risks from spills of oil and drilling fluids and cross-contamination of aquifers if well casings are not properly sealed. Abandoned wells that are not adequately capped may be used for illegal dumping that can contaminate source water.

Implications of Scientific Uncertainty

Evaluating risks associated with managing forests and grasslands that are sources of public drinking water requires close collaboration among managers, scientists, and the public. Clearly, the most current scientific findings need to be considered in this process, but the participants in this joint effort will need to recognize that science alone cannot solve many management problems. An appropriate role for science is to provide the public, in the case of government-owned land, or the responsible party, in the case of private land, with a better understanding of the effects of land-use decisions on drinking water sources.

As should be apparent from this report, there are many situations in which scientific studies provide limited or, in cases where knowledge is scant, almost no basis for evaluating risks of some activities to public water sources. To cite a few examples: the risks to source water from dispersed recreation, from the deposition of nitrogen compounds from air pollution, or from newly discovered pathogens might prove to be difficult to estimate with current knowledge. When contaminant sources are suspected, but their effects on source water are highly uncertain, it is prudent to implement backup activities such as monitoring of water quality at source water intakes. Monitoring may offer additional benefits beyond limiting the immediate risk to human health. It can also allow landowners and land managers to learn from experience; they can adjust future management practices on the basis of past results and can help to reassure the public that their water source is being protected even where predictions of land-use effects cannot be made precisely. To deal with large, uncontrolled events with severe consequences, such as floods or accidental chemical spills, emergency preparedness measures should be instituted. In this way, public water supplies can be protected by minimizing damage from such potential catastrophes.

Gaps in scientific knowledge about the effects of forest and grassland management on drinking water sources indicate areas in which future research can provide large potential benefits for land managers. Some examples emerged in the preparation of this report. There has been very little research on the risk that land management activities may introduce disease organisms into source water. Research is needed on the potential to transmit pathogens to source water from urban-wildland intermixed development, from water contact activities such as swimming as well as recreational pursuits that occur in areas without developed sanitary facilities, and from management that concentrates animals near source water. Animals that may transmit pathogens include water birds and other wildlife, as well as domestic animals such as livestock and pets. Research is needed on how land management activities such as timber and fire management in areas of high nitrogen deposition from air pollution may affect the release of nitrates into source water. There is a need for research on how forest and grassland management practices affect the economic costs and benefits of providing safe drinking water and to whom costs and benefits may accrue.

Implications for Source Water Assessments

The Forest Service and other government agencies are developing relatively new management approaches, such as ecosystem management and sustainability, to ensure the long-term viability of species and human communities that depend on public land. A key objective of these practices is to consider a broad spectrum of values in land management. Source water assessments are consistent with this approach because they increase the public's awareness of source water protection as a value to be included in land management. Teams that perform source water assessments should be composed of individuals from a wide variety of disciplines and should draw upon a wide spectrum of public opinion to ensure that the many relevant points of view are included in the analyses. In addition, these teams should make an effort to integrate across disciplines because many risks to source waters arise at the intersection of activities traditionally considered to be the subject of separate disciplines. For example, an impoundment constructed for the sole purpose of storing source water could become a source of contamination if it attracted large numbers of water birds to feed or roost on the reservoir or drew large numbers of recreationists to enjoy its shores.

Assessments should consider activities that take place in sensitive parts of the landscape that have strong linkages to surface water. Activities in these sensitive areas should be examined carefully for contaminant risk, especially during floods. Examples of areas with strong hydrological linkages to surface water are: active stream channels or lakebeds, riparian areas, floodplains, and areas near wetlands, springs, and seeps. Surface water moves quickly through the landscape. Contaminants that reach surface water can reach drinking water intakes rapidly, with relatively little time for transformation or removal by natural processes. When incidents such as chemical spills contaminate surface water, fast action may be needed to protect drinking water. Once the source of surface contamination is contained, however, the rapid movement of surface water tends to flush away contaminants quickly. Where possible, potential contaminant sources should not be permitted in these areas, and vegetated buffer strips should be retained in these zones.

Some sensitive areas may be linked by gravity to surface water. Steep slopes and areas with unstable geomorphology close to water bodies carry high risks. Activities that disturb the soil surface or remove significant portions of the vegetation in these sensitive areas may cause high erosion rates or landslides that may increase sediment loads for drinking water sources downslope.

Some sensitive zones may also have strong biological linkages to surface water. Both wild and domestic grazing animals are strongly attracted to surface water, especially in arid areas or during droughts when forage may be concentrated near water. Water bird populations and migratory fish can likewise be drawn from long distances to suitable habitat in or near water. Management actions that cause high animal concentrations in or near drinking water sources need to be carefully examined for their potential to introduce contaminants.

Some sensitive parts of the landscape have strong linkages to ground water that is used for drinking water sources. Recharge areas that have high infiltration of water to aquifers are particularly susceptible to contamination. Ground water and its contaminants usually move very slowly. This may allow long warning times for systems that rely on ground water, e.g., when contaminants are detected in ground water moving towards drinking water wells. However, once an impurity is introduced into an aquifer, contamination may be essentially permanent because ground water movement may take a very long time to flush away impurities. Exceptions exist where underground conduits such as caves, permit rapid underground flows, making predictions of underground contaminant movement very uncertain. Facilities such as septic systems, leach fields, underground fuel tanks, and dumps in sensitive aquifer recharge areas are potential contamination sources and need careful maintenance and monitoring. Illegal dumping of toxics in recharge areas may go undetected until it reaches a well, producing potentially serious consequences for health and disruption of water supply.

Not all contaminants pose the same risks. Contaminants such as toxic substances and pathogens that can cause serious health problems if they are consumed in drinking water obviously deserve a high priority in assessing risk. Contaminants that do not pose a direct threat to health but may make water less palatable, or may interfere with or increase the cost of treatment such as clean suspended sediment, color, odor, or taste should logically not receive as high a priority as health-threatening contaminants, but the economic cost of treating these nonhealth-threatening contaminants should be included in analyses.

Source Water Protection as a Priority for Land Management

In principle, providing safe drinking water to protect human health is a high priority in our society. In practice, this priority is often not well represented in land-use decisions. As was suggested in chapter 4, appropriate land-use practices that protect clean source water may be more cost-effective for society as a whole than removing pollutants after the fact. However, decisions about land uses and their effects on water are often made piecemeal and potential savings often are not realized. Source water assessments can help to forge the connection between land use and drinking water protection by better informing land managers and the public about that linkage. For society to fully benefit, other mechanisms may need to be established that more closely link the outcomes of land management to its effect on drinking water sources. In chapter 4, an example was cited of how a water utility provided an economic incentive for land managers upstream to control contaminants. Many other legal, institutional, or economic arrangements could potentially produce similar, positive effects on source water quality. Managers of land and of drinking water systems should be encouraged to cooperate to protect drinking water safety at the local level, but more fundamental and far-reaching changes may be required at the policy level to enhance incentives that encourage such cooperation and to overcome obstacles that inhibit it.

The human values that drive economic and political decisions affecting drinking water sources are reflections of the values held by the general public. Over time, the values that people place on natural resources can be expected to change and the purposes that public lands serve will likewise change. A challenge for land managers and the public will be to see that, as new patterns of land uses arise in the future, the importance of safe sources of drinking water to protect human health is given due consideration among the competing uses of forests and grasslands.

We hope that land managers, scientists, and the public will draw upon the basic scientific information in this report and apply it to their local watersheds as they participate in source water assessments in their State. We expect that they will also draw upon their own experience and use their best professional judgement to decide what portions of this work are most relevant to their particular setting within their individual source areas. In doing so, they may find important factors and interactions at work in their particular watersheds that we have overlooked, and will reveal future research needs. We look forward to the results of their analyses, because they will advance the understanding of the relationship between land management and source water quality. Their efforts will be an important next step toward better protection of drinking water sources in future land management decisions.

Literature Cited

U.S. EPA. 1993. Guidance specifying management measures for sources of nonpoint pollution on coastal waters. EPA–840–B–92–002. Washington, DC: U.S. Environmental Protection Agency. 836 p.

U.S. Geological Survey. 1999. The quality of our Nation’s water—nutrients and pesticides: Circ. 1225. Denver: U.S. Geological Survey. 82 p.

Part VII:

Appendices

Cockolds Creek, ACE Basin, South Carolina. Photo by Bill Lea

Appendix A

City of Baltimore Municipal Reservoirs, Incorporating Forest Management Principles and Practices

Robert J. Northrop[1]

Introduction

The city of Baltimore owns and operates the Loch Raven, Prettyboy, and Liberty Reservoirs, located north and northwest of the city, in the northern Piedmont region of Maryland (appendix fig. A.1). They supply water to over 1.5 million people. The reservoirs are surrounded by 17,580 acres of city-owned forest that was acquired between 1880 and 1955 to ensure control of land use in critical areas immediately adjacent to the reservoirs. Forest management on the reservoir land dates back to 1919. Following the clearing for Loch Raven and Prettyboy Reservoirs, a logging and sawmill crew was retained for forestry work, and the first professional forester was hired. This forest management program was undertaken to harvest and sell forest products while protecting the reservoir. Revenues were used for watershed enhancements, and lumber was used by the Department of Public Works in Baltimore.

In recent years, the reservoir land has also been valued as a core area for the conservation of regional biodiversity and for dispersed outdoor recreation. In 1989, concerns about timber harvesting, uncontrolled access, and a rapid increase in recreational use convinced the city to reevaluate its management practices. At the same time, the public agencies responsible for Maryland's Source Water Protection Assessment [Safe Drinking Water Act Amendments of 1996 (Public Law 104–182)] were expressing concern over the eutrophic nature of the three reservoirs and their loss of storage capacity due to sedimentation.

The watersheds, which are the primary sources of water for the reservoirs, are in Baltimore, Harford, and Carroll Counties in Maryland, as well as York County, PA. City-owned land makes up only an average of 7 percent of the total area of the watersheds draining into each reservoir. These source water drainages are part of the urbanizing and expanding Baltimore-Washington metropolitan area, which is the fourth largest in the United States. The Prettyboy and Liberty basins, however, are still rural in character with agricultural use predominant. Preserving the quality of the water that flows into the reservoirs requires careful control of sediment, as well as point- and nonpoint-source pollutants.

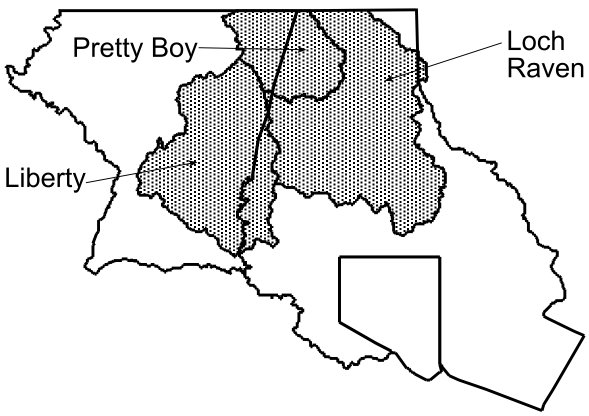
Our watershed management strategy is seen as vitally important to the continued efficient and economical provision of safe drinking water for the region's residents by all Federal, State, and local agencies. Since forest management can influence water quantity, as well as quality, by filtering and sequestering various forms of soluble and solid pollutants coming from adjacent land uses, it is recognized as a key component of the management of these watersheds. Private forest landowners are enhancing water quality by applying several forest conservation principles on their land. They are restoring forest wetlands and riparian forests and are using silvicultural practices to maintain forest vigor.

Studies

The Maryland Department of Natural Resources (MD-DNR), Forest Service, has entered into an agreement with the city of Baltimore to develop a comprehensive Forest Resource Conservation Plan for the 17,580 acres of land surrounding the Loch Raven, Prettyboy, and Liberty Reservoirs. Through a cooperative agreement with the U.S. Department of Agriculture, Forest Service, and the use of its NED–1[2] Decision Support Software, a detailed forest stand level analysis incorporating forest patch methodology will be conducted. Additional data will be collected on wildlife habitat composition and structure, and on the quality of water in first- and second-order streams. A separate recreational use survey will be conducted through contract with a regional university.

[1] Regional Watershed Forester, Maryland Department of Natural Resources–Forest Service, North East, MD.

[2] A prescription design system that incorporates management goals for multiple objectives, analyzes current forest conditions, produces ecommendations for management alternatives, and predicts future conditions under different alternatives. This system assists in evaluating silvicultural decisions at a project level using landscape-scale factors.



Appendix figure A.1—Watersheds supplying Baltimore, MD, with water.

Goals for conservation were set through a series of 20 public meetings conducted by the city of Baltimore's Department of Public Works during 1991. These goals included:

1. The protection and enhancement of water quality.
2. The maintenance and restoration of regional biological diversity within the public lands surrounding the reservoirs.
3. The management of woodlands to maximize forest habitat value.
4. Providing recreational opportunities compatible with the above objectives.

Concurrently, the MD-DNR Forest Service has also begun work with the Baltimore Metropolitan Council of Governments and the Gunpowder Watershed Project. They have a U.S. Environmental Protection Agency (EPA) small watershed grant project, where Federal, State, and local staff work to develop cooperative and collaborative strategies to address various environmental issues, including source water protection, in a holistic fashion at the watershed level. Background data and information on the Loch Raven and Prettyboy Reservoir drainage basins are being supplied through the Maryland Department of the Environment's Source Water Assessment Program (Safe Drinking Water Act, sec. 1453). Background data and information on the Liberty Reservoir drainage basin are being supplied through the Department of Natural Resources' Unified Watershed Assessment, as part of the State's Clean Water Action Plan.

Both studies used land-use loading coefficients to estimate the pounds of nutrients and sediment typically produced for a classified land use. Composite storm event samples and baseflow were collected at various sites in the Prettyboy and Loch Raven drainage basins but not the Liberty basin. Preliminary reports from the modeling exercises indicate that there is a statistically significant increasing trend in nitrate concentrations reaching the Loch Raven Reservoir, with highest concentrations in baseflow, indicating historical ground water contamination. These reports are also indicating that nutrient contamination is widespread throughout the Prettyboy and Liberty drainage basins as well, with current levels in the same range as at the Loch Raven basin.

Source water protection strategies being developed by these groups and associated State agencies highlight the need to conserve the existing forest in a healthy and vigorous condition. Forest wetlands and riparian forests need to be restored for their functional ability to filter sediment and other suspended solids, sequester pollutants in woody tissue, and promote denitrification. Forest wetland and riparian forest restoration activities within the three drainage basins will be targeted to specific sites that provide the best opportunity to intercept ground water and overland flows before they reach the receiving streams. Using a Geographic Information System (ArcView) and data layers from various Federal, State, and local agencies, the MD-DNR has developed a method that locates and ranks potential restoration sites. Potential forest wetland restoration sites are located by identifying hydric soils that lack natural vegetative cover. This system also locates potential riparian forest restoration sites by identifying stream segments that lack forest cover and assessing their potential to improve water quality. A weighed ranking is assigned based upon the nutrient loading potential of adjacent land uses, the size of the ownership parcel, and stream order (lower order streams receive higher ranking).

Interest in the management of the city-owned and surrounding forest is keen. Public support is critical to the plan's successful implementation. The Friends of the Watershed, an existing city-sponsored citizen's advisory group, will be invited to review data sets that are being collected, as well as the proposed analysis. They will also be asked to assist in the identification of public meeting sites and the context for stakeholder involvement.

Anticipated Results

Through the analysis of forest resources at multiple scales (unit to ecoregion) and timeframes, city-owned tracts will be evaluated to determine their potential to:

1. Serve as buffers to adjacent land uses.

2. Support an increasing desire on behalf of the growing urban population for outdoor recreation.

3. Assist in the conservation of biological diversity at the regional scale.

The comprehensive forest conservation plan will provide explicit management recommendations, allowing the city to plan and organize its conservation activities in the most efficient and effective manner. The deliberate and comprehensive involvement of interested citizens and community associations will lead to the public consensus the city needs to once again feel comfortable in actively managing its properties for the multiple values consistent with its stated goals.

Finally, the plan will offer forest management guidance to Federal, State, and local agencies concerned with the continued decline in the region's forest land base. This decline has been compounded by the cumulative impacts of pollution, fragmentation, and habitat loss. The plan will support and clarify the functions of forest resources as integral to the long-term sustainability of local watersheds and lead to the incorporation of forest management techniques into watershed strategies concerned with water quality.

Appendix B

Managing the Shift from Water Yield to Water Quality on Boston's Water Supply Watersheds

Thom Kyker-Snowman[1]

Boston's drinking water derives from surface reservoirs within three major watersheds: the Quabbin, Ware River, and Wachusett (appendix fig. B.1). These watersheds total in excess of 225,000 acres [90 000 hectares (ha)]; about 40 percent is under the care and control of the Metropolitan District Commission's Division of Watershed Management (MDC-DWM). This system supplies approximately 250 million gallons [900 million liters (L)] daily to accommodate the drinking water demands of 2.5 million people, about 40 percent of the population of the Commonwealth. The water is currently treated (chlorine and chloramines for disinfection, fluoride to promote healthy teeth, and soda ash and carbon dioxide to prevent corrosion of pipes), but not filtered. The objective to avoid the costs and the many other ramifications of filtration is at the center of current management decision-making for these watersheds. This objective represents a dramatic shift from the focus on water quantity, which has dominated the history of Boston's water supply.

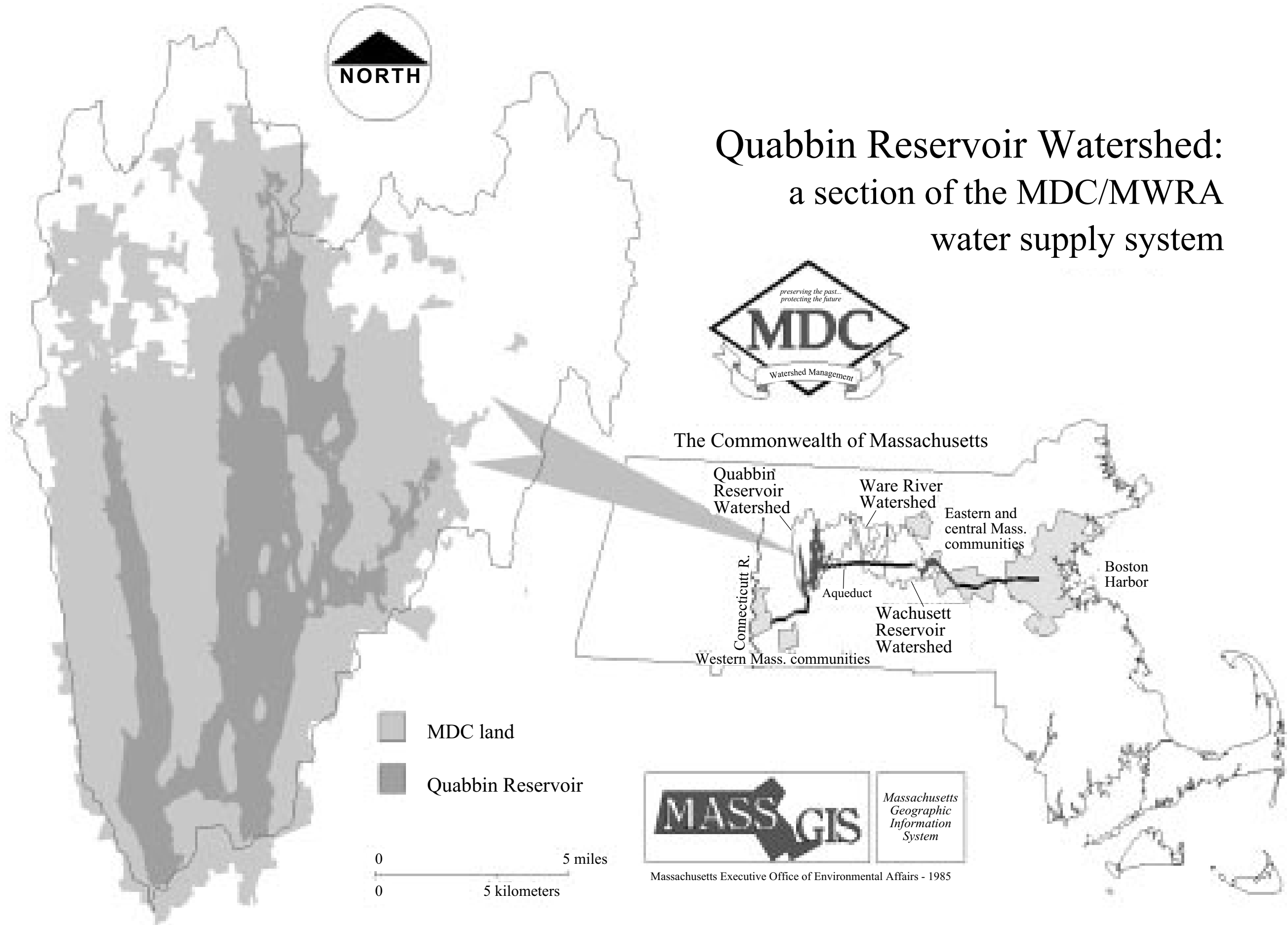
Since the settlement of Boston, its citizens continued to look west to meet the increasing demand for water. In 1795, the Aqueduct Corporation was created to tap Jamaica Pond in Roxbury to supply the 20,000 Boston inhabitants. In 1848, Lake Cochituate was added, and in short order from 1870–80, the Sudbury River and Framingham Reservoirs came on line. By 1895, Boston's population exceeded 500,000 and the metropolitan area exceeded a million. The Wachusett Reservoir, the largest reservoir in the World at the time, was built by 1908 and added a 65 billion gallon (234 billion L) capacity to the system. This was still not enough to keep up with the growing demand. Then Boston tapped the Ware River with an aqueduct to the Wachusett Reservoir, and finally constructed the 412-billion-gallon (1500-billion-L) Quabbin Reservoir within the Swift River Valley.

Despite these efforts, water quantity persisted as a concern. In 1967, just 20 years after the Quabbin Reservoir filled to capacity, a severe drought lowered the reservoir to 45 percent of its capacity and skeptics worried it would ever fill again. Although Quabbin Reservoir filled to capacity again by 1976, water demands were exceeding the safe yield from the system (300 million gallons per day or 1.1 billion L per day) by almost 50 million gallons (180 million L) per day. After lengthy debates about augmenting supplies by diverting the Connecticut or Millers River to the reservoirs, the MDC-DWM was mandated to address the situation by increasing water yield from its lands. The primary approach was to clearcut 2,000 acres (800 ha) of red pine (*Pinus resinosa* Ait.) plantations and to convert them to grass fields, which was estimated to provide an additional 300 million gallons (1.1 billion L) of water annually. After a 1989 drought dropped Quabbin Reservoir to a 17-year low, authorities declared a water emergency. Water conservation efforts (spurred in part by raising water rates), and an aggressive leak detection and repair program have dramatically lowered water consumption. Today, the daily draw on the system is 50 million gallons (180 million L) below its safe yield.

In addition to converting pine plantations to grass, MDC-DWM postponed management of an inflated deer population, because it was thought that deer browsing the understory could increase in water yield. The deer population in the Quabbin watershed had grown to nearly 48 to 80 deer per square mile (19 to 31 deer per square kilometer) (6 to 8 times the statewide average), under the hunting restrictions on MDC-DWM lands. Early forest management plans had acknowledged the impact of this population on the understory. The emphasis on water yield made it easier to choose to avoid the difficult politics associated with starting a deer management program, especially following 50 years of hunting prohibition.

Changes in drinking water laws and regulations have dramatically altered the approach to managing natural resources on the watersheds whose waters are unfiltered surface supplies. The Federal Safe Drinking Water Act (SDWA) became law in 1974, and set national standards for maximum contaminant levels and treatment techniques.

[1] Natural Resources Specialist, Massachusetts Metropolitan District Commission, Division of Watershed Management, Belchertown, MA.



Appendix figure B.1—Location of Quabbin Reservoir and water supply system for Boston, MA.

Amendments to the SDWA in 1986 established a priority for using filtration as a dominant treatment technique. The EPA addressed this priority through the Surface Water Treatment Rule of 1989 (SWTR), which essentially required that all surface water supplies be filtered unless a supply could pass a rigorous test allowing it to qualify for a waiver from filtration. The SWTR established disinfection and monitoring requirements and set new limits for pathogens and turbidity, which indicate the success or failure of either artificial or natural filtration.

It has been estimated that the construction costs alone for a filtration plant for Boston's water supply would exceed $200 million. This alone is a strong incentive to maintain a waiver, but perhaps more important is the threat of losing the mandate for watershed protection, should filtration become a reality. The MDC-DWM currently owns and controls 64 percent of the Quabbin watershed (appendix fig. B.1), and this control is a critical argument in favor of relying on natural filtration. If artificial filtration were installed, it is worth wondering if the budget required to manage MDC-DWM lands and to pay tax substitutes to the local towns would persist. Similarly, recreation is carefully limited on these watershed lands, and it would be increasingly difficult to resist these pressures in the absence of a requirement for natural filtration of Boston's drinking water.

The combination of reduced pressure to increase yields and of the increasing desire to avoid filtration have shifted the management focus in the Quabbin watershed away from water production and sharply toward water-quality protection. From the natural resources perspective, this meant demonstrating that wildlife and forest are being managed to avoid degrading and, if possible, improving the natural filtration process. Two major wildlife issues were met squarely along these lines: water birds (in particular, gulls and geese) and white-tailed deer. Seagulls threaten the maintenance of water-quality standards when they spend their days feeding in landfills and returning to roost by the thousands on open surface water supplies, transporting pathogens that can threaten human health. The MDC-DWM has devised an elaborate gull-harassment program that deals with the problem by moving roosting birds far from the water supply intakes.

The browsing by the high populations of white-tailed deer consumes the forest understory and threatens the regeneration of forest cover if it is lost to natural or human disturbance. The threat of major overstory losses associated with catastrophic hurricanes can recur in New England every 100 to 150 years; the most recent was in 1938. A model to predict hurricane damage was developed by Harvard University on their forest in Petersham, MA, immediately adjacent to the Quabbin Reservation. This model predicted in 1992 that 50 to 75 percent of the conifers and 25 to 75 percent of the hardwoods in the Quabbin watershed would be damaged by such a storm. During the writing of the most recent Quabbin land management plan, it was decided that an even-aged, relatively mature forest with greatly impaired regenerative capacity was incompatible with the desire to maintain predictable long-term natural filtration of the drinking water supply.

The first step in reversing this untenable condition was to reduce the impact of deer, primarily through controlled hunting. The MDC-DWM engaged in a lengthy, multiyear public campaign for support of this idea, which overcame opposition including a Federal lawsuit filed by an animal-rights organization. This suit claimed that there was a probability that an unrecovered deer, wounded by a lead slug, would die and be fed upon by a bald eagle, which might in turn ingest lead from the wound and die as a result. At that time, this would have constituted an illegal taking of a Federally protected, endangered species. The plan to reinstitute hunting, in order to protect the drinking water supply, persisted through this debate, and the first hunt in 50 years was conducted in 1991. Hunting has continued since then, and regeneration of both trees and other understory plants has been dramatic as a result. Wildflowers like trillium (*Trillum* spp.) and marsh-marigold (*Caltha* spp.) that were not found before hunting have reappeared after a long absence.

In addition, plans called for diversifying both the age and the species structure of the watershed forest cover. This objective calls for maintaining an understory as the reserve forest; a midstory for its rapid nutrient uptake; and an overstory for its regulation of organic decomposition, its provision of seed, and the water infiltration and retention function of its deep root system. These canopy layers are to be balanced, in an uneven-aged silvicultural approach, throughout the managed forest surrounding MDC-DWM reservoirs. This deliberate restructuring is accomplished through commercial harvesting using primarily group selection and irregular shelterwood approaches. The drinking water supply context mandates state-of-the-art best management practices, including a requirement that all equipment be supplied with a spill kit for potential oil leaks and strict restrictions on ground pressures allowed on sensitive land.

The working hypothesis of this approach is that frequent, endogenous disturbance of the scale of group-selection silviculture will lessen the amplitude of the disturbance wave represented by infrequent, exogenous disturbances, such as catastrophic hurricanes. The MDC-DWM made the commitment that any short-term negative effects of timber harvesting would not exceed the long-term benefits to drinking water derived from this deliberate forest structuring. While the large volume of Quabbin Reservoir dilutes differences in tributary water quality, the no-net-negative policy will require intensive monitoring at the tributary level, especially during storm events and spring runoff. This monitoring effort has recently begun at Quabbin and will hopefully quantify the effects of incorporating large, infrequent disturbances into management planning for unfiltered surface supplies of drinking water.

Appendix C

Cumulative Impacts of Land Use on Water Quality in a Southern Appalachian Watershed[1]

Wayne T. Swank and Paul V. Bolstad[2]

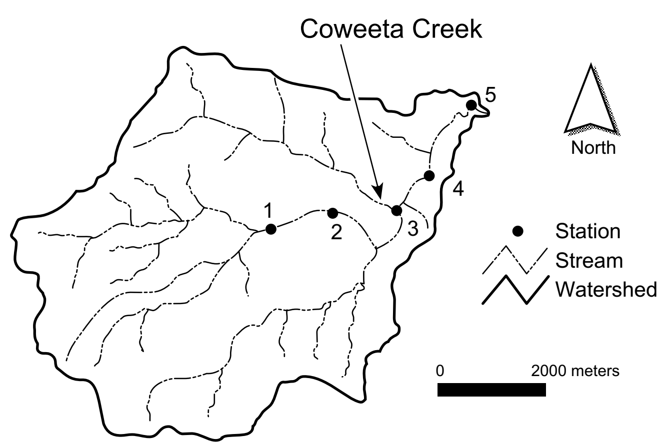
Introduction

Water-quality variables were sampled over 109 weeks along Coweeta Creek, a fifth-order stream located in the Appalachian Mountains of western North Carolina. The purpose of the study was to observe any changes in water quality over a range of flow conditions with concomitant downstream changes in the mix of land uses. Variables sampled include pH, bicarbonate (HCO_3^{-1}), conductivity, nitrate nitrogen (NO_3-N), ammonium nitrogen (NH_4^+-N), phosphate phosphorus (PO_4^{3-}—P), chloride (Cl^{-1}), sodium (Na^+), potassium (K^+), calcium (Ca^{2+}), magnesium (Mg^{2+}), sulfate (SO_4^{-2}), silica (SiO_2), turbidity, temperature, dissolved oxygen, total and fecal coliform, and fecal streptococcus. Landcover and land use or both were interpreted from 1:20,000 aerial photographs and entered in a Geographic Information System, along with information on total and paved road length, building location and density, catchment boundaries, hydrography, and slope. Linear regressions were performed to related basin and near stream landscape variables to water quality.

Five water-quality monitoring stations were located over 5.4 miles (8.7 kilometers) of Coweeta Creek (appendix fig. C.1). Along Coweeta Creek, stream size and permanent landscape alteration increases, e.g., conversion of forest to agriculture and increases in road density, from lower to higher station numbers (appendix table C.1). Sites were selected to encompass incremental additions and a variety of land uses. Most of the area above station 1 was covered with mature, deciduous forest, and paved road density was low, while unpaved road density was relatively high. Downstream stations were selected to encompass additional land-use features such as residences along the stream, grazing and other agricultural practices, plus additional roads. Stations 2 through 4 were characterized by a 6- to 20-foot (2- to 6-meter) wide riparian shrub strip [chiefly alder, (*Alnus* spp.), bramble (*Rubus* spp.), and willow *Salix* spp.)] with a mix of pastures, homesites, and farmland beyond the riparian strip. Station 5 was in a low-density suburban mix, with mown grass to the stream edge.

Streamwater samples were collected during baseflow and stormflow periods. During baseflow, grab samples were collected in 1-liter bottles from the free-flowing section of the stream. Sampling was initiated the first week of June 1991 and was conducted twice weekly through August. Thereafter, baseflow sampling was conducted approximately weekly through the first week of November 1993.

During selected storm events, two different sampling methods were used. Grab samples were taken on the rising limb of the hydrograph, near peak flow, and on the hydrograph recession. Some storm events were also sampled using a time-proportional automated sampler, which was activated near storm onset.



Appendix figure C.1—Watershed boundary and stream sampling locations in the Coweeta Creek Watershed in western North Carolina. Stations 1 through 5 are arranged down the stream gradient on Coweeta Creek. First-order streams are not shown.

[1] This example is excerpted from a paper by Bolstad and Swank (1997).
[2] Retired Project Leader, USDA Forest Service, Southern Research Station, Otto, NC; and Professor of Forestry, University of Minnesota, Minneapolis, MN, respectively.

Appendix table C.1—Summary data for the catchments above five sampling stations along Coweeta Creek in western North Carolina

Characteristics upstream of sample station	Units	Sampling station number: Upstream 1	2	3	4	Downstream 5
Total area	Ha	1605	1798	3099	4163	4456
Forest area	Ha	1600	1782	2986	3904	4113
Agricultural area	Ha	4	13	89	155	192
Urban/suburban area	Ha	1	3	24	104	151
Total road length	Km	39.8	45.2	80.8	106.8	122.6
Unpaved road length	Km	38.6	43.9	73.4	96.4	106.5
Total road density	Km/km^2	2.49	2.51	2.61	2.60	2.75
Unpaved road density	Km/km^2	2.41	2.44	2.37	2.33	2.39
Structures/area	No./100 ha	.37	3.06	5.36	6.01	9.23

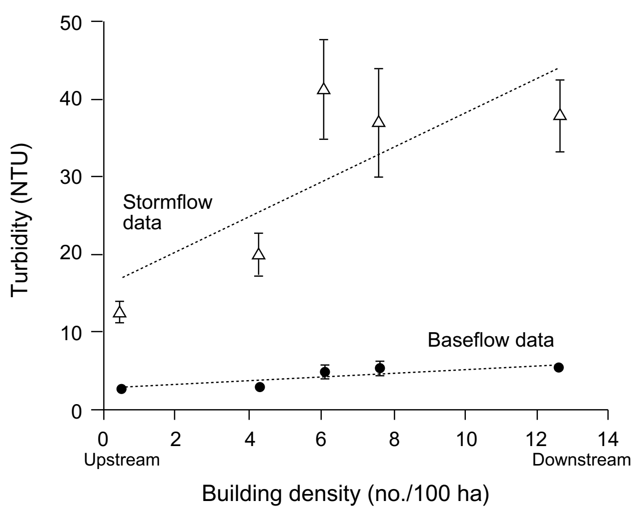
Baseflow Water Quality

Water quality was good during baseflow conditions over the 3-year study period. Concentrations of most solutes averaged < 1 milligram per liter, typical of stream chemistry for lightly disturbed forest watersheds in the Southern Appalachians. NO_3-N, NH_4^+-N, and PO_4^{3-}—P were very low, indicating the absence of point sources of inorganic solutes into the stream. Turbidity during baseflow was generally low, typical for the Southern Appalachians (appendix fig. C.2), averaging < 6 nephalometric turbidity units for all stations. Mean counts of total fecal coliform and fecal streptococci at station 1 were typical of mean values reported for other streams draining relatively undisturbed forested watersheds in western North Carolina. Several variables showed distinct downstream increases. Cation concentrations, SiO_2, HCO_3^{-1}, SO_4^{-2}, Cl^{-1}, conductivity, turbidity, and temperature generally increased downstream from station 1 to 5.

Mean baseflow levels for total coliform, fecal coliform, and streptococci counts increased from threefold to eightfold downstream (appendix table C.2). Thus, there is a cumulative increase in bacteria populations, indicating additive sources downstream. The transport of these bacteria is probably primarily through the soil or direct input by warm-blooded vertebrates, e.g., raccoons, livestock, since base-flow samples represent periods when there is little or no overland flow input from adjacent lands.

Stormflow Water Quality

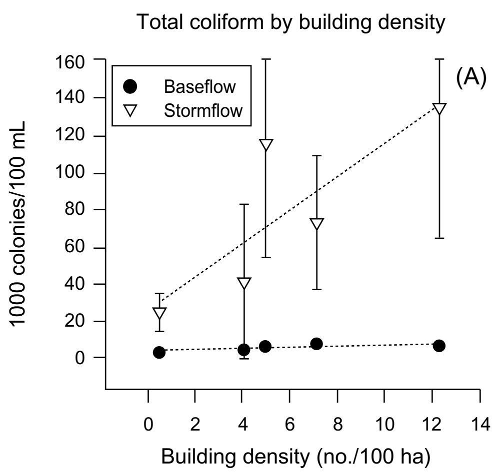
Conductivity, NO_3-N, HCO_3^-, Cl-, K^+, Na^+, Ca^{2+}, Mg^{2+}, SiO_2, turbidity, temperature, and total coliform often showed cumulative increases downstream. Two patterns were obvious in comparing stormflow and baseflow data. First, mean values for most variables at most stations were higher during stormflow. These increases range from slight and nonsignificant to quite large (turbidity, appendix fig. C.2).

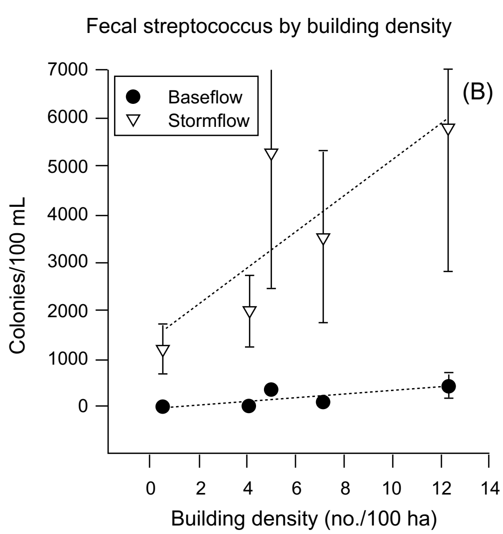


Appendix figure C.2—Mean and standard error (bars) for turbidity, plotted against building density for each sampling condition (baseflow and stormflow).

Appendix table C.2—Summary water-quality data from baseflow grab samples (means) at each of the five sampling stations along Coweeta Creek in western North Carolina

Variable	Station number 1	2	3	4	5
	Per 100 milliliters				
Total coliform	9,470	13,660	40,040	30,740	52,140
Fecal coliform	200	340	460	1,130	840
Fecal streptococcus	710	1,310	2,180	1,590	1,840





Appendix figure C.3—Mean and standard error (bars) for (A) total coliform and (B) fecal streptococcus, plotted against building density for each sampling condition (baseflow and stormflow). (Building density increases downstream).

Appendix table C.3—Summary water-quality data from stormflow samples (means) at each of the five sampling stations along Coweeta Creek in western North Carolina

Variable	Station number 1	2	3	4	5
	Per 100 milliliters				
Total coliform	18,790	34,640	NA	77,160	98,390
Fecal coliform	880	130	NA	970	1,260
Fecal streptococcus	450	8,710	NA	3,260	4,190

NA = Not available.

Bacteria levels were among the most responsive water-quality variables during storm events, although patterns were highly variable among storms and among seasons. Total coliform, fecal coliform, and fecal streptococci typically increased twofold to threefold during storm events compared to baseflow populations. The source of these large downstream increases in bacteria may be attributed to observed overland flow from adjacent lands directly into streams, disturbance of bottom sediments, and streambank flushing (appendix table C.3, appendix fig. C.3).

Conclusions

In summary, this work identifies consistent, cumulative downstream changes in Coweeta Creek concomitant with downstream changes in land use. Furthermore, this work indicates consistently higher downstream changes during stormflow when compared to baseflow conditions, suggesting cumulative impacts due to landscape alteration, as tested here, are much greater during stormflow events.

Literature Cited

Bolstad, P.V.; Swank, W.T. 1997. Cumulative impacts of landuse on water quality in a Southern Appalachian watershed. Journal of the American Water Resources Association: 519–533. Vol. 33, no. 3.

Appendix D

Protozoan Pathogens *Giardia* and *Cryptosporidium*

David Stern[1]

Introduction

Two pathogens carried by wildlife, *Giardia* spp. and *Cryptosporidium* spp., are of great interest in drinking water. *Giardia* cysts and *Cryptosporidium* oocysts are parasitic protozoans. They are active and reproduce within their hosts and encyst to survive in the environment during transmission between hosts. Many species of wildlife have been found to be hosts for these parasites (appendix table D.1). These organisms are significant sources of gastrointestinal illness (Jokipii and others 1983, Kenney 1994, Moore and others 1994). The risk posed by these parasites is believed to be significant. As little as one cyst may be able to cause infection (Medema and others 1995, Rose and Gerba 1991, Rose and others 1991). What is more, these organisms are resistant to disinfection (Campbell and others 1982, Clark and Regli 1993, Craun 1981, Haas and Heller 1990, Hoff and Rubin 1987, Jarroll and others 1981, Kong and others 1988, Quinn and Betts 1993, Rice 1981, Rubin and others 1989). Although there are medications that are effective in treating giardiasis, currently there are no drugs available to treat infections caused by *Cryptosporidium.*

Federal Regulation

In 1986, the U.S. Congress recognized the threat posed by these protozoan parasites and revised the Safe Drinking Water Act to begin to address this concern. The Surface Water Treatment Rule for the Safe Drinking Water Act (SDWA), promulgated in 1991 (U.S. Environmental Protection Agency 1996), requires all surface water supplies to be filtered prior to distribution to the public, unless it can be demonstrated that a certain level of purity exists and can be maintained. The Surface Water Treatment Rule emphasizes water supply filtration because disinfecting by chlorination does not eliminate the threat posed by *Giardia* and *Cryptosporidium* (Campbell and others 1982, Clark and others 1989). Accordingly, prevention or filtration is recommended as the response to this threat. In its continuance of concern for the threat posed by *Giardia* and *Cryptosporidium*, the Federal Government enacted 1996 amendments to the SDWA to fund additional watershed research on these organisms.

[1] Pathogen Program Supervisor, New York City, Department of Environmental Protection, Valhalla, NY.

Giardia

To survive in the environment, *Giardia* encysts itself into a resistant form. *Giardia* cysts are 5 to 15 microns in size and oblong in shape. Early research on this parasitic protozoan identified it on the basis of median body morphology and the host it was found in. Accordingly, *Giardia* species are *G. muris, G. agilis,* and *G. duodenalis* and are usually found in rodents, frogs, and warm-blooded vertebrates, respectively. This early nomenclature was due to the assumption that *Giardia* was highly host-specific. More recent research has shown that *Giardia* can cross-infect different species of hosts (Meyer and Jarroll 1980).

Identification of *Giardia* as a waterborne parasite for humans was first reported in the 1940's during a study of a disease outbreak in an apartment building in Tokyo, Japan (Davis 1948). *Giardia* has more recently been reported as the most frequently identified parasite responsible for disease outbreaks in surface water supplies in North America (Craun 1984). A significant portion of the literature has reported on the occurrence, disinfection, and treatment of *Giardia* cysts.

The life cycle for *Giardia* has been described by Meyer and Jarroll (1980). *Giardia* is monoxenous, which means that all of its life stages occur in one host. These stages include an inactive cyst form that is capable of resisting environmental stresses and a free-living form known as a trophozoite. The trophozoite has a ventral sucker disk that attaches to the intestinal wall to obtain subsidence. The life cycle consists of: (1) a host ingesting the cyst, (2) excystation (emergence of the trophozoite out of the cyst) occurring in the small intestine after the cyst has been subjected to the digestive environment, (3) the released trophozoite attaching to the intestinal wall where it feeds and reproduces by binary fission, and (4) some of the reproduced trophozoites encyst within the intestine and the resultant cyst is excreted in the infected animal's feces to be transmitted to other hosts.

Cryptosporidium

Cryptosporidium was first identified by Tyzzer (1910) over 90 years ago as a parasite of the common mouse. Its

Appendix table D.1—Species reported as hosts to protozoans *Giardia* and *Cryptosporidium*

Species	Common name	Parasites hosted
Pisces		
Cyprinus carpio	Carp	*Cryptosporidium*
Sciaenops ocellatus	Red drum	*Cryptosporidium*
Plecostomus spp.	Catfish	*Cryptosporidium*
Salmo trutta	Brown trout	*Cryptosporidium*
Oncorhynchus mykiss	Rainbow trout	*Cryptosporidium*
Amphibia		
Ceratophrys ornata	Bell's horned frog	*Cryptosporidium*
Bufo americanus	American toad	*Cryptosporidium*
B. regularis	Common toad	*Giardia*
Rana pipiens	Leopard frog	*Giardia*
R. clamitans	Green frog	*Giardia*
Reptilia		
Chelonia mydas	Green turtle	*Cryptosporidium*
Geochelone elegans	Star tortoise	*Cryptosporidium*
G. carbonaria	Red-footed tortoise	*Cryptosporidium*
Squamata Lacertilia (lizards)		
Agama aculeata	Kalahari spiny agama	*Cryptosporidium*
A. planiceps	Damara rock agama	*Cryptosporidium*
Chameleo c. senegalensis	Chamelon	*Cryptosporidium*
Chamaeleo pardalis	Panther chamelon	*Cryptosporidium*
Chlamydosaurus kingi	Frilled lizard	*Cryptosporidium*
Lacerta lepida	Ocellated lacerta	*Cryptosporidium*
Chondrodactylus angulifer	Sand gecko	*Cryptosporidium*
Serpentes (snakes)		
Crotalus durissus culminatus	Rattlesnake	*Cryptosporidium*
Sistrurus miliarius	Pygmy rattlesnake	*Cryptosporidium*
Lampropeltis getulus holbrooki	Say's kingsnake	*Giardia*
Elaphe subocularis	Trans-Pecos rat snake	*Cryptosporidium*
E. o. obsoleta	Black rat snake	*Cryptosporidium*
E. o. quadrivittata	Yellow rat snake	*Cryptosporidium*
E. o. lindheimeri	Texas rat snake	*Cryptosporidium*
E. guttata	Corn snake	*Cryptosporidium*
E. v. vulpina	Western fox snake	*Cryptosporidium*
Gonysoma oxycephala	Red-tailed green rat snake	*Cryptosporidium*
Pituophis melanoleucus	Black pine snake	*Cryptosporidium*
P. melanoleucus catenifer	Gopher snake	*Cryptosporidium*
Drymarchon corais couperi	Eastern indigo snake	*Cryptosporidium*
Lampropeltis zonata pulchura	San Diego mountain snake	*Cryptosporidium*
L. triangulum	Various subspecies	*Cryptosporidium*
Nerodia h. harteri	Brazos water snake	*Cryptosporidium*
N. r. rhombifera	Diamondback water snake	*Cryptosporidium*
Boiga dendrophila	Mangrove snake	*Cryptosporidium*
C. horridus	Timber rattlesnake	*Cryptosporidium*
C. atrioca udatus	Canebrake rattlesnake	*Cryptosporidium*
C. l. lepidus	Rock rattlesnake	*Cryptosporidium*

continued

Appendix table D.1—Species reported as hosts to protozoans *Giardia* and *Cryptosporidium* (continued)

Species	Common name	Parasites hosted
Aves		
Anseriformes		
Branta canadensis	Canada goose	*Cryptosporidium, Giardia*
Anser anser	Domestic goose	*Cryptosporidium*
Cygnus spp.	Tundra swan	*Cryptosporidium*
C. olor	Mute swan	*Cryptosporidium*
Aix sponsa	Wood duck	*Cryptosporidium*
Anas platyrrhynchos	Mallard duck	*Cryptosporidium*
Mergus merganser	Common merganser	*Cryptosporidium*
Columbiformes		
Columba livia	Pigeon	*Cryptosporidium*
Galliformes		
Gallus gallus	Chicken	*Cryptosporidium*
Meleagris gallopavo	Turkey	*Cryptosporidium*
Coturnix coturnix	Common quail	*Cryptosporidium*
Colinus virginianus	Bobwhite quail	*Cryptosporidium*
Phasianus colchicus	Ring-necked pheasant	*Cryptosporidium*
Pavo cristatus	Peafowl	*Cryptosporidium*
Perdix perdix	Grey partridge	*Cryptosporidium*
Alectoris graeca	Chuckar partridge	*Cryptosporidium*
Numida meleagris	Guinea fowl	*Cryptosporidium*
Charadriiformes		
Larus ridibundus	Black-headed gull	*Cryptosporidium*
L. argentatus	Herring gull	*Cryptosporidium*
L. delawarensis	Ring-billed gull	*Cryptosporidium*
Recurvirostra avosetta	Avocet	*Giardia*
Threskiornis spinicollis	Straw-necked ibis	*Giardia*
Passeriformes		
Poephila cincta	Black-throated finch	*Cryptosporidium*
Lonchura cucullata	Bronze mannikin finch, red cheek finch	*Cryptosporidium*
Passer domesticus	House sparrow	*Giardia*
Zonotrichia georgiana	Swamp sparrow	*Giardia*
Sturnella neglecta	Western meadowlark	*Giardia*
Lanius collurio	Red-backed shrike	*Giardia*
Ciconiiformes		
Ardea herodias	Great blue heron	*Giardia*
A. cinerea	Gray heron	*Giardia*
A. cocoi	Cocoi heron	*Giardia*
Egretta alba	Great egret	*Giardia*
E. caerulea	Little blue heron	*Giardia*
Nycticorax nyctocorax	Black-crowned night-heron	*Giardia*
N. naevius	Night-heron	*Giardia*
Butorides virescens	Green-backed heron	*Giardia*
Egretta intermedia	Intermediate egret	*Giardia*
Bubulcus ibis	Cattle egret	*Giardia*
Botaurus lentiginosus	American bittern	*Giardia*
Ixobrychus minutus	Little bittern	*Giardia*
Plegadis falcinellus	Glossy ibis	*Giardia*

continued

Appendix table D.1—Species reported as hosts to protozoans *Giardia* and *Cryptosporidium* (continued)

Species	Common name	Parasites hosted
Aves (continued)		
Falconiformes		
Cathartes aura	Turkey vulture	*Giardia*
Elanus caeruleus	Black-winged kite	*Giardia*
Mammalia		
Marsupialia		
Didelphis virginiana	Virginia opossum	*Cryptosporidium, Giardia*
Pseudocheirus peregrinus	Possum	*Giardia*
Insectivora		
Sorex spp.	Shrew	*Giardia*
Blarina brevicauda	Short-tailed shrew	*Cryptosporidium*, *Giardia*
S. cinereus	Masked shrew	*Cryptosporidium*
Parascalops breweri	Hairy-tailed mole	*Cryptosporidium*
Myotis lucifugus	Little brown bat	*Cryptosporidium*
Eptesicus fuscus	Big brown bat	*Cryptosporidium*, *Giardia*
Lagomorpha		
Sylvilagus floridanus	Eastern cottontail	*Cryptosporidium*, *Giardia*
Rodentia		
Ondatra zibethica	Common muskrat	*Cryptosporidium*, *Giardia*
Microtus agrestis	Field vole	*Cryptosporidium*
M. chrotorrhinus	Rock vole	*Giardia*
M. pennsylvanicus	Meadow vole	*Cryptosporidium*, *Giardia*
M. pinetorum	Pine vole	*Giardia*
M. longicaudus	Long-tailed vole	*Giardia*
M. ochrogaster	Prairie vole	*Giardia*
M. californicus	Meadow vole	*Giardia*
M. richardsoni	Water vole	*Giardia*
Clethrionomys glareolus	Bank vole	*Cryptosporidium*
C. glareolus skomerensis	Skomer bank vole	*Cryptosporidium*
Apodemus sylvaticus	Wood mouse	*Cryptosporidium*
Rattus rattus	Roof rat or ship rat	*Cryptosporidium, Giardia*
Sigmodon hispidus	Cotton rat	*Cryptosporidium*
Erithizon dorsatum	Porcupine	*Cryptosporidium*, *Giardia*
Mus musculus	House mouse	*Cryptosporidium*, *Giardia*
Zapus hudsonicus	Meadow jumping mouse	*Giardia*
Napaeozapus insignis	Woodland jumping mouse	*Giardia*
Peromyscus leucopus	White-footed mouse	*Cryptosporidium*, *Giardia*
P. maniculatus	Deer mouse	*Cryptosporidium*, *Giardia*
C. gapperi	Red-backed vole	*Cryptosporidium*, *Giardia*
Pitymys savii	Savi's woodland vole	*Giardia*
R. norvegicus	Norway rat	*Cryptosporidium*, *Giardia*
Neotoma cinerea	Wood rat	*Giardia*
Dipodomys heermanni	Kangaroo rat	*Giardia*
Tamia striatus	Eastern chipmunk	*Cryptosporidium*, *Giardia*
Sciurus carolinensis	Eastern gray squirrel	*Cryptosporidium*, *Giardia*
Tamiasciurus hudsonicus	Red squirrel	*Cryptosporidium*, *Giardia*
Glaucomys volans	Southern flying squirrel	*Giardia*

continued

Appendix table D.1—Species reported as hosts to protozoans *Giardia* and *Cryptosporidium* (continued)

Species	Common name	Parasites hosted
Mammelia (continued)		
Rodentia (continued)		
Spermophilus beecheyi	Ground squirrel	*Giardia*
S. richardsoni	Richardson's ground squirrel	*Giardia*
S. tridecemlineatus	13-lined ground squirrel	*Giardia*
Marmota monax	Woodchuck	*Cryptosporidium*, *Giardia*
Coendu villosus	Tree porcupine	*Giardia*
Carnivora		
Ursus americanus	Black bear	*Cryptosporidium*
Mustela erminea	Short-tailed weasel	*Cryptosporidium*
M. putorius furo	Ferret	*Cryptosporidium*
Canis latrans	Coyote	*Cryptosporidium*, *Giardia*
Vulpes vulpes	Red fox	*Giardia*
Urocyon cineroargenteus	Gray fox	*Cryptosporidium*, *Giardia*
Procyon lotor	Raccoon	*Cryptosporidium*, *Giardia*
Paradoxurus h. hermaphroditus	Palm civet	*Giardia*
Mephitis mephitis	Striped skunk	*Cryptosporidium*, *Giardia*
Mustela vision	Mink	*Cryptosporidium*, *Giardia*
M. nigripes	Black-footed ferret	*Giardia*
Meles meles	Badger	*Giardia*
Lynx rufus	Bobcat	*Cryptosporidium*, *Giardia*
Sirenia		
Dugong dugong	Manatee	*Cryptosporidium*
Ruminants		
Cervus canadensis	Elk, wapiti	*Giardia*
Odocoileus virginiana	White-tailed deer	*Cryptosporidium*, *Giardia*
Antilocapra americana	Pronghorn	*Giardia*
Ovis canadensis x. O. musimon	Bighorn x mouflon sheep	*Giardia*
Llama glama	Llama	*Cryptosporidium, Giardia*
Odocoileus hemionus	Mule deer	*Cryptosporidium*

Source: Adapted from Wade and others, in press.

significance began to be recognized in the 1970's when a number of reports identified *Cryptosporidium* as the cause of diarrhea in calves (O'Donoghue 1995). Human infections began to be reported in the mid-1970's and by the 1990's, *Cryptosporidium* was recognized as a significant threat to individuals that are immunocompromised (Current and Garcia 1991, Ungar 1990).

Cryptosporidium oocysts are spherical and 4 to 6 microns in diameter (Barer and Wright 1990, Casemore 1991, Casemore and others 1985, Current 1987, Current and others 1986, Fayer and Ungar 1986, Issac-Renton and others 1987, O'Donoghue 1995, Smith and Rose 1990, Ungar 1994). Most oocysts contain up to four sporozoites (free-living form). A number of species have been identified among various hosts. Many of these species can cross-infect different species of hosts. Several *Cryptosporidium* species are found more often in association with certain host species, especially when the host species are vertebrates. Thus, *C. muris* is common in mammals, *C. meleagridis* in birds, *C. crotalia* in reptiles, and *C. nasorum* in fish (Levine 1984, O'Donoghue 1995).

O'Donoghue (1995) and Current and Bick (1989) described the life cycle for *Cryptosporidium*. Like *Giardia*, *Cryptosporidium* encysts to survive outside its host, and its life stages occur in the infected animal. Its life cycle is more complex due to the addition of a sexual stage of reproduction within the host. The oocyst of *Crypto-sporidium* undergoes excystation (release of sporozoites) after it has been ingested by a host and has been subjected to conditions usually found in a digestive system. These conditions have been identified as including temperature, low pH, and digestive enzymes (Fayer and Leek 1984, Reduker and Speer 1985). The released sporozoites attach to epithelial cells of the small intestine, where they are identified as trophozoites (*Cryptosporidium* attached to intestine). The trophozoites mature into meronts that produce merozoites through asexual reproduction. The merozoites, in turn, develop into either other meronts or produce the sexual form of *Cryptosporidium*, microgametes (male form) and macrogametes (female form). The mobile microgametes fertilize the macrogametes in sexual reproduction to form a zygote (the sexually reproduced form of *Cryptosporidium*). Most of the zygotes form thick-walled oocysts that are released from the host to infect other hosts and complete the life cycle.

Literature Cited

Barer, R.R.; Wright, A.E. 1990. *Cryptosporidium* and water. Applied Microbiology. 11(6): 271–277.

Campbell, I.; Tzipori, S.; Hutcinson, G.; Angus, K.W. 1982. Effects of disinfection on survival of *Cryptosporidium* oocysts. Veterinary Record. 111(18): 414–415.

Casemore, D.P. 1991. Epidemiology of human Cryptosporidiosis and the water route of infection. Water Science and Technology. 24(2): 157–164.

Casemore, D.P.; Sands, R.L.; Curry, A. 1985. *Cryptosporidium* species a " new " human pathogen. Journal of Clinical Pathology. 38(12): 1321–1336.

Clark, R.M.; Read, E.J.; Hoff, J.C. 1989. Analysis of inactivation of *Giardia lamblia* by chlorine. Journal of Environmental Engineering. 115(1): 80–90.

Clark, R.M.; Regli, S. 1993. Development of *Giardia* Ct values for the surface water treatment rule. Journal of Environmental Science and Health. 28(5): 1081–1097.

Craun, G.F. 1981. Disease outbreaks caused by drinking water. Journal of the Water Pollution Control Federation. 53(6): 1134–1138.

Craun, G.F. 1984. Waterborne outbreaks of giardiasis. Current status. In: Erlandsen, S.L.; Meyer, E.A., eds. *Giardia* and giardiasis. New York: Plenum Press: 243–262.

Current, W.L. 1987. *Cryptosporidium*: its biology and potential for environmental transmission. CRC Critical Reviews in Environmental Control. 17(1): 21–51.

Current, W.L.; Bick, P. 1989. Immunobiology of *Cryptosporidium* spp. Pathology and Immunopathology Research. 8: 141–160.

Current, W.L.; Garcia, L.S. 1991. Cryptosporidiosis. Clinical Microbiology Reviews. 4: 325–358.

Current, W.L.; Upton, S.J.; Haynes, T.B. 1986. The life cycle of *Cryptosporidium baileyi* n.sp. (*Apicomplexa*, *Cryptosporidiidae*) infecting chickens. Journal of Protozoology. 33: 29–296.

Davis, C.; Ritchie, L.S. 1948. Clinical manifestations and treatment of epidemic amebiasis occurring in occupants of the Mantetsu apartment building. Tokyo: American Journal of Tropical Medicine. 28: 817–823.

Fayer, R.; Leek, R.G. 1984. The effects of reducing conditions, medium, pH, temperature, and time on in vitro excystation of *Cryptosporidium*. Journal of Protozoa. 31: 567–569.

Fayer, R.; Ungar, B.L.P. 1986. *Cryptosporidium* spp. and Cryptosporidiosis. Microbiol.Reviews. 50(4): 458–483.

Haas, C.N.; Heller, B. 1990. Kinetics of inactivation of *Giardia lamblia* by free chlorine. Water Research. 24(4): 233–238.

Hoff, J.C.; Rubin, A.J. 1987. Use of inactivation rate data for establishing drinking water disinfection requirements for *Giardia* cyst control. In: Preprints of papers presented at the 194th American chemical society national meeting; [Date of meeting unknown]; Washington, DC. Washington, DC: American Chemical Society, Division of Environmental Chemistry: 760–763. Vol. 27, no.2.

Issac-Renton, J.L.; Fogel, D.; Stibbs, H.H.; Ongerth, J.E. 1987. *Giardia* and *Cryptosporidium* in drinking water. The Lancet. April: 973–974.

Jarroll, E.L.; Bingham, A.K.; Meyer, E.A. 1981. Effect of chlorine on *Giardia lamblia* cyst viability. Applied and Environmental Microbiology. 41(2): 483–487.

Jokipii, L.; Pohjola, S.; Jokipii, A.M.M. 1983. *Cryptosporidium*: a frequent finding in patients with gastrointestinal symptoms. The Lancet. 2(8346): 358–842.

Kenney, R.T. 1994. Parasitic causes of diarrhea. Pediatric Annals. 23(8): 414–421.

Kong, L.I.; Swango, L.J.; Blagburn, B.L. [and others]. 1988. Inactivation of *Giardia lamblia* and *Giardia canis* cysts by combined and free chlorine. Applied and Environmental Microbiology. 54(10): 2580–2582.

Levine, N.D. 1984. Taxonology and review of the Cooidian Genus *Cyrptosporidium* (Protozoa, Aploomiexa). Journal of Protozoology. 31: 94–98.

Medema, G.J.; Van Asperen, I.A.; Klokman Houweling, J.M. [and others]. 1995. The relationship between health effects in triathletes and microbiological quality of freshwater. Water-Science-Technology. 31(5–6): 19–26.

Meyer, E.A.; Jarroll, E.L. 1980. *Giardia* and giardiasis. American Journal of Epidemiology. 111: 1–12.

Moore, A.C.; Herwaldt, B.L.; Craun, G.F. [and others]. 1994. Waterborne disease in the United States, 1991 and 1992. Journal American Water Works Association. February: 87–99.

O'Donoghue, P.J. 1995. *Cryptosporidium* and Cryptosporidiosis in man and animals. International Journal for Parasitology. 25(2): 139–195.

Quinn, C.M.; Betts, W.B. 1993. Longer term viability status of chlorine-treated *Crypyosporidium* oocysts in tap water. Biomedical Letter. 48: 315–318.

Reduker, D.W.; Speer, C.A. 1985. Factors influencing excystation of *Cryptosporidium* oocysts from cattle. Journal of Parasitology. 71(1): 112–115.

Rice, G.R. 1981. Use of ozone in drinking water treatment. Journal American Water Works Association. 73(1): 44.

Rose, J.B.; Gerba, C.P. 1991. Use of risk assessment for development of microbial standards. Water Science and Technology. 24(2): 29–34.

Rose, J.B.; Haas, C.N.; Regli, S. 1991. Risk assessment and control of waterborne Giardiasis. American Journal of Public Health. 81(6): 709–713.

Rubin, A.J.; Evers, D.P.; Eyman, C.M.; Jarroll, E.L. 1989. Inactivation of gerbil-cultured *Giardia lamblia* cysts by free chlorine: Applied and Environmental Microbiology. 55(10): 2592–2594.

Smith, H.V.; Rose, J.B. 1990. Waterborne Cryptosporidiosis. Parasitology Today. 6(1): 8–12.

Tyzzer, E.E. 1910. An extracellular coccidium, *Cryptosporidium muris*, of the gastric glands of the common mouse. Journal of Medical Research. 23: 487–509.

Ungar, B.L.P. 1990. Cryptosporidiosis in humans (*Homo sapiens*). Boston: CRC Press: 59–82.

Ungar, B.L.P. 1994. Cryptosporidium and Cryptosporidiosis. Textbook of aids medicine. Baltimore: Williams and Wilkins: 323–343.

Wade, S.E.; Ziegler, P.E.; Schaff, S.L. [In press]. Preliminary investigation of *Giardia* species and *Cryptosporidium* species in wildlife in the New York City watershed. Final report. Ithaca, NY: Cornell University, College of Veterinary Medicine, Department of Population Medicine and Diagnostic Science.

Appendix E

Water Treatment Technologies Tables

Gary Logsdon[1]

Most raw water is not suitable for human consumption without treatment. Some water only needs to be filtered and disinfected before consumption (Committee on Small Water Supply Systems, National Research Council 1997). Other water must be treated with additional processes to remove specific chemical contaminants or nuisance chemicals like iron and manganese. Appendix tables E.1 to E.4 present information on water treatment techniques that can be used for controlling common contaminants. The tables provide guidance on selecting the appropriate treatment processes. However, a water treatment specialist must select the best process on a site-specific basis. Additional information can be found in recent volumes of Water Quality and Treatment (Letterman 1999) and Safe Water from Every Tap (Committee on Small Water Supply Systems, National Research Council 1997).

Literature Cited

Committee on Small Water Supply Systems, National Research Council. 1997. Safe water from every tap: improving water service to small communities. Washington, DC: National Academy Press. 230 p.

Letterman, R.D., tech. ed. 1999. Water quality and treatment. 5th ed. New York: McGraw-Hill, Inc. 1,248 p.

[1] Director, Black and Veatch, Water Process Research, Cincinnati, OH.

Appendix table E.1—Water treatment technologies by disinfectants, oxidants, and aeration[a]

General water quality-constituent	Free chlorine	Chloramine	Chlorine dioxide	Ozone	Potassium permanganate	Ultraviolet radiation	Aeration
General water qualities							
Turbidity, sediment							
Color	x		x	x	x		
Disinfection by-product precursors							
Taste and odor			x	x	x	x	x
Biological contaminants							
Algae							
Protozoa	x		x	x		x	
Bacteria	x	x	x	x		x	
Viruses	x	x	x	x		x	
Organic chemicals							
Volatile organics							x
Semi-volatile compounds							
Pesticides and herbicides							
Biodegradable organic matter							
Inorganic chemicals							
Hardness							
Iron[b]	x		x	x	x		x
Manganese[b]	x		x	x	x		
Arsenic							
Selenium							
Thallium							
Fluoride							
Radon							x
Radium							
Uranium							
Cations							
Anions							
Total dissolved solids							
Nitrate							
Ammonia							

[a] The columns and rows lacking x's are where process is not appropriate or recommended for the constituent.
[b] When oxidant is followed by filtration.
Source: Table adapted from Committee on Small Water Supply Systems, National Research Council 1997.

Appendix table E.2—Water treatment technologies by type of adsorption and ion exchange system[a]

General water quality-constituent	Powdered activated carbon	Granulated activated carbon	Ion exchange	Activated alumina
General water qualities				
Turbidity, sediment				
Color	x	x		
Disinfection by-product precursors	x	x		
Taste and odor	x	x		
Biological contaminants				
Algae		x		
Protozoa		x		
Bacteria		x		
Viruses		x		
Organic chemicals				
Volatile organics	x	x		
Semi-volatile compounds	x	x		
Pesticides and herbicides	x	x		
Biodegradable organic matter	x	x		
Inorganic chemicals				
Hardness			x	
Iron				
Manganese				
Arsenic				x
Selenium				x
Thallium				x
Fluoride			x	x
Radon				
Radium			x	
Uranium			x	
Cations			x	
Anions			x	
Total dissolved solids				
Nitrate			x	
Ammonia				

[a] The columns and rows lacking x's are where process is not appropriate or recommended for the constituent.
Source: Table adapted from Committee on Small Water Supply Systems, National Research Council 1997.

Appendix table E.3—Water treatment technologies by type of membrane treatment system[a]

General water quality-constituent	Microfiltration	Ultrafiltration	Nanofiltration	Reverse osmosis	Electrodialysis/ ED reversal
General water qualities					
Turbidity, sediment	x	x	x		
Color		x	x	x	
Disinfection by-product precursors		x	x	x	
Taste and odor					
Biological contaminants					
Algae	x	x	x		
Protozoa	x	x	x	x	
Bacteria		x	x	x	
Viruses			x	x	
Organic chemicals					
Volatile organics					
Semi-volatile compounds				x	
Pesticides and herbicides			x	x	
Biodegradable organic matter					
Inorganic chemicals					
Hardness			x	x	x
Iron					x
Manganese					x
Arsenic				x	x
Selenium				x	x
Thallium				x	x
Fluoride				x	x
Radon					
Radium				x	x
Uranium				x	x
Cations				x	x
Anions				x	x
Total dissolved solids				x	x
Nitrate				x	x
Ammonia					

ED = electrodialysis.
[a] The columns and rows lacking x’s are where process is not appropriate or recommended for the constituent.
Source: Table adapted from Committee on Small Water Supply Systems, National Research Council 1997.

Appendix table E.4—Water treatment technologies by type of filtration system[a]

General water quality-constituent	Direct filtration	Conventional filtration	Dissolved air flotation	Precoat filtration	Slow sand filtration	Bag/cartridge filters	Lime softening
General water qualities							
Turbidity, sediment	x	x	x	x	x		x
Color	x	x	x				
Disinfection by-product precursors	x	x	x				
Taste and odor					x		
Biological contaminants							
Algae		x	x	x			
Protozoa	x	x	x	x	x	x	x
Bacteria	x	x	x	x	x		x
Viruses	x	x	x	x	x		x
Organic chemicals							
Volatile organics							
Semi-volatile compounds							
Pesticides and herbicides							
Biodegradable organic matter	x*	x*	x*		x		
Inorganic chemicals							
Hardness							x
Iron	x	x	x	x			x
Manganese	x	x	x	x			x
Arsenic		x	x				x
Selenium							x
Thallium							
Fluoride							
Radon							
Radium							x
Uranium							
Cations							x
Anions							
Total dissolved solids							
Nitrate							
Ammonia	x*	x*	x*		x		

X* = when the filter is operated in a biologically active mode.
[a] The columns and rows lacking x's are where process is not appropriate or recommended for the constituent.
Source: Table adapted from National Research Council 1997.

List of Figures

Page

Page

List of Tables

Page

Page

Glossary of Abbreviations and Acronyms

ai: active ingredient

aum: animal unit month

BAER: Burn Area Emergency Rehabilitation

BLM: Bureau of Land Management

BMP: best management practice

BOD: biological oxygen demand

Bt: *Bacillus thuringiensis*

BTEX: benzene, toluene, ethylbenzene, xylene

C: carbon

Ca: calcium

Ca^{2+}: calcium ion

$Ca(HCO_3)_2$: calcium bicarbonate

Cd: cadmium

CDC: Centers for Disease Control and Prevention

CERLA: Comprehensive Environmental Response, Compensation and Liability Act

CH_4: methane

Cl: chlorine

Cl^{-1}: chloride ion

cm: centimeter

CO_2: carbon dioxide

COD: chemical oxygen demand

Cu: copper

CWD: coarse woody debris

DEP: City of New York Department of Environmental Protection

DO: dissolved oxygen

DOD: Department of Defense

EPA: U.S. Environmental Protection Agency

ET: evapotranspiration

F: fluorine

FC: fecal coliform

FDA: U.S. Food and Drug Administration

Fe^{+2}: ferrous iron ion

Fe^{+3}: ferric iron ion

Fe_3S_4: greigite

FeS: pyrite

FeS_2: marcasite

FIFRA: Federal Insecticide, Fungicide, and Rodenticide Act

FS: fecal streptococcus

GIS: Geographic Information System

H^+: hydrogen ion

H_2S: hydrogen sulfide

ha: hectare

HA or HAL: health advisory level

HCN: hydrogen cyanide

HCO_3^{-1}: bicarbonate ion

HTH: chlorine

JTU: Jackson turbidity unit

K: potassium

K^+: potassium ion

km: kilometer

LOAEL: lowest observed adverse effect level

m: meter

MCL: maximum contaminant level

MCLG: maximum contaminant level goal

MD-DNR: Maryland Department of Natural Resources

MDC-DWM: Metropolitan District Commission's Division of Watershed Management (Massachusetts)

Mg: magnesium

Mg^{2+}: magnesium ion

mg: milligram

Mg: metric tonne or megagram

µg: microgram

µm: micron or 1 millionth of a meter

$Mg(HCO_3)_2$: magnesium bicarbonate

$MgSO_4$: magnesium sulfate

MITC: methyl isothiocyanate

Mn: manganese

MTBE: methyl tertiary butyl ester

N: nitrogen

N_2: nitrogen gas

N_2O: nitrous oxide

Na: sodium

Na^+: sodium ion

NAWQA: National Water-Quality Assessment Program

NEPA: National Environmental Policy Act

NFS: National Forest System

NH_4^{+1}: ammonium ion

NH_4+-N: ammonium-nitrogen

NO_2^{-1}: nitrite ion

NO_3^{-1}: nitrate ion

NO_3-N: nitrate-nitrogen

NOAEL: no observed adverse effect level

NOEL: no observed effect level

NTU: nephalometric turbidity unit

O: oxygen

O_2: oxygen gas

OPS: Office of Pipeline Safety

OSM: Office of Surface Mining Reclamation and Enforcement

P: phosphorus

PAH: polycyclic aromatic hydrocarbon

Pb: lead

PFC: Proper Functioning Condition

pH: the negative logarithm (base 10) of the hydrogen ion concentration

PO_4^{-3}: phosphate ion

PO_4^{3-}—P: phosphate phosphorus

ppb: parts per billion

ppm: parts per million

RCRA: Resource Conservation and Recovery Act

RfD: reference dose

S: sulfur

SDWA: Safe Drinking Water Act Amendments of 1996

Si: silicon

SiO_2: silica

SMRCA: Surface Mining and Control Reclamation Act

SO_4^{-2}: sulfate ion

SWA: source water assessment

SWAP: source water assessment program

SWTR: Surface Water Treatment Rule of 1989

TDS: total dissolved solids

TFM: 3-trifluoromethyl-4-nitrophenol

THM: trihalomethane

TT: treatment technique

USGS: U.S. Geological Survey

UST: underground storage tanks

VOC: volatile organic compound

WATSED: Region 1 water and sediment model

WEPP: Water Erosion Prediction Project

WPP: Watershed Protection Program

Zn: zinc

Glossary of Terms

action level: the level of contamination which, if exceeded, triggers treatment or other requirements that a water system must follow.

acute health effect: an immediate, i.e., within hours or days, effect that may result from exposure to certain drinking water contaminants.

allelopathy: a chemical defense mechanism in certain plants to keep other plants from growing under or around their canopy.

anions: negatively charged ions. The most common in natural waters are bicarbonate, nitrate, sulfate, chloride, and different forms of phosphorus.

anoxic: lacking oxygen.

aquifer: a saturated, permeable geologic unit that can transmit significant quantities of water under ordinary conditions.

aquitard: a geologic unit that cannot transmit significant quantities of water under ordinary conditions.

artesian well: a deep well in which water rises under pressure from a permeable strata.

bed load: bed load is sediment too heavy to be continuously suspended in flowing water. This material is rolled or bounded along the stream bottom. The size of particles making up the bed load varies with streamflow, velocity, particle density and shape, and many other factors.

biological oxygen demand (BOD): dissolved oxygen required to decompose biodegradable organic material in parts per million (ppm) or milligrams per liter (mg/L).

capillary fringe: the zone between the water table and the vadose zone where water is held within pores by capillary forces.

cations: positively charged ions. The most common in natural waters are calcium, sodium, potassium, and ammonium.

centralized wastewater treatment system: water treatment system that collects wastewater and transports wastewater via sewers to a central treatment facility.

chemical oxygen demand (COD): dissolved oxygen required to decompose biodegradable and nondegradable organic material in parts per million (ppm) or milligrams per liter (mg/L).

closed well: a well that has been permanently disconnected and capped or filled so that contamination cannot move from the surface into the aquifer.

coliform: a group of related bacteria whose presence in drinking water may indicate contamination by disease-causing microorganisms.

coliphages: viruses (bacteriophage) that infect and replicate in the bacterium *E. coli* and appear to be present wherever *E. coli* are found. Some strains are more resistant to chlorine disinfection than total coliforms.

commercial use: includes water for motels, hotels, restaurants, office buildings, golf courses, civilian and military institutions, and in some areas fish hatcheries. The consumptive use of water for commercial purposes in the United States in 1995 was estimated at 14 percent of withdrawals and deliveries.

community water system: a water system with 15 or more service connections and which supplies drinking water to 25 or more of the same people year-round in their residences.

compliance: the act of meeting all Federal and State drinking water regulations.

confined aquifer: an aquifer that is between two impermeable geologic units.

consumptive use: the part of water withdrawn that is evaporated, transpired, or incorporated into products or crops. In many instances, the consumptive use is the difference between the amount delivered and the amount released.

contaminant: any substance found in water (including microorganisms, minerals, chemicals, radionuclides, etc.) which may be harmful to human health.

conveyance loss: the quantity of water that is lost in transit from its source to point of use or point of return.

***Cryptosporidium* spp.:** a microorganism commonly found in lakes and rivers which is highly resistant to disinfection. *Cryptosporidium* has caused several large outbreaks of gastrointestinal illness, with symptoms that include diarrhea, nausea, and/or stomach cramps.

cutslope: excavated slope uphill from a road located on the side of a steep hill.

decentralized water treatment system: onsite water treatment facility.

disinfectant: a chemical (commonly chlorine, chloramine, or ozone) or physical process, e.g., ultraviolet light, that kills microoganisms such as bacteria, viruses, and protozoa.

distribution system: a network of pipes leading from a treatment plant to customer's plumbing system.

domestic use: includes water used for normal household purposes, such as drinking, food preparation, bathing, washing clothes and dishes, flushing toilets, and watering lawns and gardens. The consumptive use of water for domestic purposes in the United States in 1995 was estimated at 26 percent of withdrawals and deliveries.

drywell: a well used for disposal of liquid wastes, other than an improved sinkhole or subsurface fluid distribution system, completed above the water table so that its bottom and sides are typically dry except when receiving fluids.

enteric viruses: viruses which infect the gastrointestinal tract of humans and are excreted with the feces of the infected individual. These viruses are excreted in relatively large numbers from infected individuals and include polioviruses, coxsackieviruses, echoviruses, other enteroviruses, adenoviruses, rotaviruses, hepatitis A virus, Norwalk viruses, astrovirus, and caliciviruses.

equivalents per liter: a chemical term indicating the number of moles of solute multiplied by the valence of the solute species in 1 liter of solution.

eutrophication: enrichment of surface waters with nutrients, especially phosphorus and nitrogen that leads to enhanced plant growth, algal blooms and depleted oxygen levels as this plant material decays.

exotic: with reference to vegetation, refers to nonnative plant species introduced either accidentally, or to meet some management goal.

fillslope: the downhill embankment on a road constructed on the side of a steep hill.

finished water: drinking water that has been treated and is ready to be delivered to customers.

fire intensity: in a wildfire or prescribed burn, a qualitative term describing the rate of heat release, related to flame length.

fire severity: in a wildfire or prescribed burn, a qualitative term describing the extent of fire effects on ecosystem components, such as vegetation or soils.

gaining stream: a stream that receives flow from ground water discharge.

***Giardia* spp.:** a microorganism frequently found in rivers and lakes, which, if not treated properly, may cause diarrhea, fatigue, and cramps after ingestion.

ground water: the water that drinking water systems pump and treat from aquifers.

industrial use: includes water use for processing, washing, and cooling in facilities that manufacture products like steel, chemicals, paper, and petroleum refining. The consumptive use of water for industrial purposes in the United States in 1995 was estimated at 15 percent of total withdrawals and deliveries.

in holdings: land parcels contained within public lands that are not owned by the agency managing public lands.

inorganic contaminants: mineral-based compounds, such as metals, nitrates, and asbestos. These contaminants are naturally occurring in some water, but can also get into water through farming, chemical manufacturing, and other human activities.

instream use: water use that takes place without the water body being diverted or withdrawn from surface or ground water sources. Examples include hydroelectric power generation, navigation, freshwater dilution of saline estuaries, maintenance of minimum streamflows to support fish and wildlife habitat, and wastewater assimilation.

irrigation use: includes all water artificially applied to farm and horticultural crops and in some cases golf courses. Of the water withdrawn for irrigation in the United States in 1995, 19 percent was lost in conveyance, 61 percent was consumptive use, and 20 percent was returned to surface of ground water supplies.

karst: geologic formation in limestone strata containing numerous dissolved, undergound channels resulting in high hydraulic conductivity and high risk of ground water pollution.

leachate: a liquid, often containing extremely high concentrations of organic and inorganic pollutants, formed from the decomposition of municipal solid waste.

livestock use: includes offstream use of water for livestock, feed lots, dairies, fish farms, and other on-farm needs. The consumptive use of water for livestock in the United States in 1995 was estimated at 26 percent of withdrawals and deliveries.

losing stream: a stream that loses water to the ground water. Streams that help recharge ground water.

mass concentration: the mass of a solute dissolved in a specific unit volume of solution, usually expressed in milligrams per liter.

mass failure: the collapse of a steep embankment when the gravitational forces within the embankment exceed the strength of the soil to maintain the current slope.

maximum contaminant level (MCL): the highest level of a contaminant that U.S. Environmental Protection Agency allows in drinking water to ensure that drinking water does not pose either a short-term or long-term health risk.

maximum contaminant level goal (MCLG): the level of a contaminant at which there would be no risk to human health. This goal is not always economically or technologically feasible, and the goal is not legally enforceable.

microorganisms: tiny living organisms that can be seen only with the aid of a microscope. Some microorganisms can cause acute health problems when consumed in drinking water. Also known as microbes.

mining use: offstream water uses for the extraction and milling of naturally occurring minerals including coals and ores, petroleum, and natural gases. The consumptive use of water for mining purposes in the United States in 1995 was estimated at 27 percent of withdrawals and deliveries.

molality: the number of moles of solute in a 1 kilogram mass of solvent.

molarity: the number of moles of solute in a liter of solution.

monitoring: testing that water systems must perform to detect and measure contaminants. A public water system that does not follow U.S. Environmental Protection Agency's monitoring methodology or schedule is in violation, and may be subject to legal action.

municipal solid waste landfill: a discrete area of land or an excavation that receives household wastes.

nonpoint-source pollution: contaminants that come from diffuse sources and pollute surface and ground water sources.

nontransient, noncommunity water system: a system which supplies drinking water to 25 or more of the same people at least 6 months per year in places other than their residences. Some examples are schools, factories, office buildings, and hospitals that have their own water systems.

offstream use: water that is diverted or withdrawn from surface or ground water sources and conveyed to the place of use.

organic contaminants: carbon-based chemicals, such as solvents and pesticides, which can get into water through runoff from cropland or discharge from factories. U.S. Environmental Protection Agency has set legal limits on 50 organic contaminants.

oxygenates: organic compounds added to gasoline to increase oxygen content of gasoline and reduce certain emissions.

pathogen: a disease-causing organism.

perched water table: a zone of saturation that is bound below by impermeable material elevated above a vadose zone above the water table.

pH: a common measure of acidity and alkalinity defined as the negative logarithm (base 10) of the hydrogen ion concentration. A pH of 7 represents neutral conditions, a pH value < 5 indicates moderately acidic conditions, while a pH value > 9 indicates moderately alkaline conditions.

plugging: the act or process of stopping the flow of water, oil, or gas into or out of a formation through a borehole or well penetrating that formation.

point-source pollution: contaminants that can be traced to specific points of discharge and pollute surface and ground water sources.

polycyclic aromatic hydrocarbons (PAH): multiple-ringed carbon compounds that are potentially carcinogenic.

potentimetric surface: the water surface level of the saturated zone in a confined aquifer.

public supply: water withdrawn by public and private suppliers and delivered to multiple users for domestic, commercial, industrial, and thermoelectric power uses. The difference between the amount of water withdrawn and delivered to users typically represents losses in the distribution system and use for water treatment plant filter cleaning, water for fire fighting, street cleaning, and occasionally municipal buildings.

public water system: any water system which provides water to at least 25 people for at least 60 days.

radionuclides: any man-made or natural element that emits radiation and that may cause cancer after many years of exposure through drinking water.

raw water: water in its natural state, prior to any treatment for drinking.

redox potential: a measure of the oxidizing or reducing capacity of a solution where positive values indicate oxidizing tendencies and negative values indicate reducing tendencies. Chemically, it is defined as the energy gained in the transfer of 1 mole of electrons from an oxidant to hydrogen.

return flow: the quantity of water that is discharged to a surface or ground water after release from the point of use and, thus, becomes available for further uses.

road prism: the road and surrounding area directly influenced by the road, including any cutslopes, ditches, fullslopes, and the roadway.

roadway: the surface of a road on which vehicles travel.

rock buttress: a thick layer of rock placed on top of a steep sideslope to reduce the risk of a mass failure.

saturated zone: a soil or geologic zone in which all pores are filled with water.

secondary drinking water standards: nonenforceable Federal guidelines regarding cosmetic effects (e.g., tooth or skin discoloration) or aesthetic effects (e.g., taste, odor, or color) of drinking water.

slumping: a mass failure generally due to an increase in water content within the soil profile on steep slopes.

sole-source aquifer: an aquifer that supplies 50 percent or more of the drinking water of an area.

spring: places on the land surface where the water table or an aquifer intersects the land surface, discharging ground water.

surface water: the water in sources open to the atmosphere, such as rivers, lakes, and reservoirs.

suspended sediment: suspended sediment is material light enough to be carried in suspension in streamflow. The sediment carried in suspension may be either organic or inorganic material. Unless specified, both types are included in suspended sediment estimates. Suspended sediment is often reported as the concentration in water using parts per million or milligrams per liter interchangeably to express the instantaneous concentration at a given point. Sediment not transported in suspension is called bed load.

thermoelectric power: includes offstream uses for the generation of electric power with fossil fuels, nuclear, or geothermal power. In the United States in 1995, surface water supplied more than 99 percent of the thermoelectric withdrawals. Consumptive use was about 2 percent of withdrawals.

total dissolved soils (TDS): determined by weighting the solid residue obtained by evaporating a measured volume of filtered water. Reported in mass per-unit volume, typically in milligrams per liter.

total sediment yield: total sediment yield includes both suspended sediment yield and bed load yield at a point along a stream.

transient, noncommunity water system: a water system which provides water in a place such as a gas station or campground where people do not remain for long periods of time. These systems do not have to test or treat their water for contaminants that pose long-term health risks because fewer than 25 people drink the water over a long period. They still must test their water for microbes and several chemicals that pose short-term risks.

turbidity: turbidity of water is the degree to which light penetration is impeded by suspended material. Turbidity is expressed either in Jackson turbidity units (JTU) or nephelometric turbidity units (NTU).

turbidity unit (tu): one tu is the interference in the passage of light caused by a suspension of 1 milligram per liter of silica. Turbidity ≤ 5 tu is generally not noticeable to the average person.

unconfined aquifer: or water table aquifer. An aquifer in which the water table forms the upper boundary.

vadose zone: this is a geologic or soil zone, which is not saturated. It is a zone of aeration, and water in this zone follows the laws of soil physics. Water in this zone does not flow to a well.

volatile organic compounds (VOC): organic compounds that volatilize at room temperatures.

vulnerability assessment: an evaluation of drinking water source quality and its vulnerability to contamination by pathogens and toxic chemicals.

wastewater release: includes the disposal of water conveyed through a sewer system. In the United States in 1995, approximately 2 percent of these releases were reclaimed for beneficial uses, such as irrigation of golf courses and public parks.

water bar: a ditch excavated across a road to route water from the road surface or uphill ditch to a downhill ditch or hillside, to reduce surface erosion by concentrated flow, and distribute surface runoff along a hillside.

water birds: water birds refer to all waterfowl including swans, geese, and ducks (whistling ducks, marsh ducks, diving ducks, stiff-tailed ducks, and mergansers) or duck-like swimming birds including gulls, cormorants, grebes, loons, coots, and wading birds, such as herons and egrets.

water delivery: the quantity of water delivered to a specific point of use.

water release: the quantity of water released to surface water or ground water after a specific use.

water table: the level at which water stands in a well and is the point where fluid pressure in the pores is exactly atmospheric. Also called phreatic surface.

watershed: the land area from which water drains into a stream, river, lake, or reservoir.

well: a bored, drilled or driven shaft, or a dug hole, whose depth is greater than its largest surface dimension.

wellhead protection area: the area surrounding a drinking water well or well field, which is protected to prevent contamination of the well(s).

withdrawal: the quantity of water diverted or withdrawn from a surface or ground water source.

Subject Index

Dissmeyer, George E., ed. 2000. Drinking water from forests and grasslands: a synthesis of the scientific literature. Gen. Tech. Rep. SRS–39. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 246 p.

This report reviews the scientific literature about the potential of common forest and grassland management to introduce contaminants of concern to human health into public drinking water sources. Effects of managing water, urbanization, recreation, roads, timber, fire, pesticides, grazing, wildlife and fish habitat, and mineral, oil, and gas resources on public drinking water source quality are reviewed. Gaps in knowledge and research needs are indicated. Managers of national forests and grasslands and similar lands in other ownerships, environmental regulators, and citizens interested in drinking water may use this report for assessing contamination risks associated with land uses.

Keywords: Economics, nutrients, pathogens, sediments, source water assessments, toxic chemicals.