Effects of Forest and Grassland Management On Drinking Water Quality for Public Water Supplies:

A Review And Synthesis of the Scientific Literature

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Review Draft

Table of Concents

The list of figures and tables will be developed and included in the published document.

Chapter 1

Introduction

Doug Ryan & Steve Glasser

The Importance of Safe Public Drinking Water

The United States Congress justified passing the Safe Drinking Water Amendments (SDWA) of 1996 (P. L. 104-182) by stating "safe drinking water is essential to the protection of public health". For 50 years the basic axiom for public health protection has been safe drinking water reduces infectious disease and extends life expectancy (*cf.* AWWA 1953). Although most United States citizens take safe 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 public water systems.

To strengthen that process, SDWA mandates that greater protection and information be provided for the 240 million Americans served by public water supplies. Under provisions calling for stronger source water protection by the year 2003, all states are required to submit to the Environmental Protection Agency (EPA) a program for delineating source water areas for all public water supply systems, and for assessing the vulnerability of these source waters to contamination. The law requires that results of the completed source water assessments (SWA's) be made available to the public.

The source water protection strategy for ensuring safe drinking water was choisen because of its high potential to be cost-effective. A poor source of water can substantially increase treatment costs to make water drinkable. When water is so contaminated that treatment is not feasible, developing alternative water supplies can be expensive and cause delays in providing safe, affordable water. By delineating areas to protect source water and inventorying potential contamination causes will help communities know the threats to their drinking water. Then, they can efficiently and effectively 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 forest and grassland yield water relatively low in contaminants. We estimate that at least 3,400 public water systems currently depend on National Forest System watersheds. In addition, National Forests and Grasslands have over 3000 public water supplies for their campgrounds and administrative centers. National forests and grasslands provide water to 60 million people, or one fourth of the people served by public water supplies nationwide. Sixty percent of forests are non-National Forest, thus the number of people served by forests and grasslands is much greater.

With the large number of public water supplies on forests and grasslands, it is likely that many forest and grassland managers will be involved planning, implementing, or reacting to public concerns related to SWA's. The managers involvement in this process will probably vary from place to place depending on the requirements of each states, the degree of public attention to particular management activities, and the potential of specific land uses to affect source waters. At the time of writing this document, it is difficult to predict to what degree managers may become involved with this process. It is our assumption that current scientific knowledge presented in a useful form will help managers to protect the safety of drinking water sources and be better informed participants in the SWA process.

The Purpose and Scope of This Document

This document's purpose is to assist forest and grassland managers comply with the SDWA by providing them with a review and synthesis of current scientific literature concerning the effects of managing these lands on the source waters for public drinking water supplies. Its audience includes managers of National Forests and Grasslands as well as managers of other forest and grass land ownerships with similar land uses, such as other federal, state, and tribal agencies, industry and private. Federal, state, and local water quality and public water supply agencies, and community groups concerned with drinking water may also find this document useful. Regardless of ownership and where land uses are similar, the chemical, physical, biological and ecological effects on source water quality are comparable.

The book's focus is restricted to potential contamination of source water by ordinary land uses in national forests and grasslands, because the EPA and the states will decide detailed delineation criteria. We chose conventional land uses on national forests and grasslands because they clearly come under the mandate of the Forest Service, the principal sponsor of this document, and because a significant fraction of public water supplies depend on national forests and grasslands for source water. The book does not address large urban developments, large industrial complexes, row crop agriculture and concentrated animal feeding operations because they are under the oversight of other agencies with strong mandates in those areas such as the EPA or Natural Resource Conservation Service (NRCS). We did include grazing and land uses that occur where urban areas border or intermix with forests and grasslands in or near national forests and frequently raise significant concerns. We focus on issues for public water supplies, because only public supplies are examined in SWA's.

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 for classifying geographic, climatic and zones with similar characteristics into ecoregions are in use by the scientific and land management communities. But no standard system of ecosystem classification has been endorsed across relevant scientific disciplines nor among federal agencies at this time. For this reason, we cited the ecoregion information contained in the literature.

How to Use this Document

This document is intended to be used by managers as a reference for assessing effects and planning land management practices to enhance the quality of drinking water sources. When managers are concerned with the potential impact 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 estimates of the effects of a particular practice. What is unknown is important, because it may indicate what 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. There is risk in using tacit assumptions in decisions, because they can lead to unintended consequences. When scientific data is used in decision making, it has the advantage that many of the important conditions affecting 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 reduce the risk of unexpected outcomes because important causal factors and assumptions may be better known.

The document format dictated that the information had to be succinctly summarized to cover subjects broadly rather than in depth. When managers need to go more deeply into a topic, they should use the scientific literature cited as an entry point into the larger body of knowledge. Wherever possible, the cited scientific information is peer-reviewed and published. If information has not been peer-reviewed and published, it is clearly indicated in the text. Case studies presented in sidebars are meant to illustrate the complexity of actual management situations and are not necessarily based on peer-reviewed literature.

To synthesis the scientific information into a form that answers questions relevant to managers required the authors to 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 itself 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, synthesis and conclusions, and the authors have considered and responded to the comments.

Although this document is separate chapters by types of land use, we recognize that in practical situations most effects on source waters result from the cumulative effects of multiple land uses that often overlap in space and time. To address this issue we direct readers to Chapters 2 that covers the natural processes of watersheds that overlay all land uses, and to Chapter 12 that summarizes the cumulative effects of multiple land uses distributed over space and time.

In this document, we concentrate on issues that arise from the manager's need to comply with the Safe Drinking Water Act Amendments of 1996 (P. L. 104-182). This is only one of the many policies and laws that currently govern the actions of National Forest and Grassland managers. We mention here a few examples of the other principal laws that affect management, but do not treat them in detail in this document. For a more complete listing and text of these laws see USDA Forest Service (1993). A provision of the Organic Act of 1897 (30 Stat. 11) that established the National Forests "for the purpose of securing favorable conditions of water flows", has been interpreted to authorize managing these lands for water resources. Administration of national forests is currently guided primarily by four laws: the Multiple Use-Sustained Yield Act (P. L. 86-517), the National Environmental Policy Act (P. L. 91-190), the Forest and Rangeland Renewable Resources Planning Act (P. L. 93-378), and the National Forest Management Act (P. L. 94-588). Forest and grassland managers also must comply with many environmental statutes including: the Endangered Species Act (P. L. 93-205), the Clean Water Act (P. L. 80-845), and the Clean Air Act (P. L. 84-159). Activities of the Forest Service with state and private land owners was authorized by the Cooperative Forestry Assistance Act (P. L. 95-313) and amended in the 1990 Farm Bill (P. L. 101-624). The Forest and Rangeland Renewable Resources Act (P. L. 93-378), with amendments in the 1990 Farm Bill (P. L. 101-624), provided authority for research by the Forest Service. Over time the laws and policies that guide land use have changed in response changes in perceived public needs and will probably continue to do so 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. This document does not cite or endorse specific BMP's but rather presents scientific evidence to serve as a basis for developing more effective practices to protect water sources.

Some laws and prudent practice require environmental monitoring to assess the outcomes of land management. The broad topic of monitoring is beyond the scope of our effort, but implicit throughout this document is the assumption that monitoring should be an integral part of land management. The availability of scientific evidence does not eliminate all risks of unforeseen outcomes and in cases where scientific studies are lacking, risks are likely to be higher. Monitoring land use practices helps to ensure public health and other important values are indeed being protected. The scientific evidence presented provides basic information on potential effects of management, risks of contamination of source water, and designing and implementing monitoring programs.

This document focuses narrowly on protecting human health by protecting drinking water. We acknowledge that managers must consider a wider range of values in most land use decisions. It is not our intent to tell managers how to weigh a spectrum of values nor how to decide among them. Rather we wish to inform managers about the specific values related to drinking water, so that they can better take these values into considerations when they make land use decisions.

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Chapter 2

Biogeochemical and Ecosystem Level Processes

F.N. Scatena, G.S. Logsdon, et al.

Introduction

Watersheds are topographically defined areas drained by a system of connecting stream channels that discharge water, sediment and dissolved materials by an ephermeral stream to a large river. Watersheds are commonly classified by physiography (i.e. headwater, steeplands, lowland, coastal plains), environmental condition (i.e. pristine, degraded, etc.), or their principal use or land cover (i.e. forest, urban, agricultural, municipal water supply, etc.).

Municipal watersheds are managed to sustain the quantity and quality of water to minimize treatment, environmental, and economic costs. Ultimately, the quality and cost of domestic water depends on the interaction between watershed, treatment, and distribution systems (Fig. 2.1). This chapter is an overview of chemical and physical processes affecting the quantity and quality of water from the landscape until it is delivered consumers. The chemical and physical characteristics of water related to watershed management, human health, and water treatment, hydrologic and nutrient budgets, and chemical transformations are discussed. A list of terms and chemical units commonly used in water quality is tabulated in Table 2.1.

Chemical, Physical, and Biological Properties of Water

Water quality (WQ) reflects measurable physical, chemical and biological characteristics of water for a specific use (i.e. human consumption, manufacturing, agriculture, the maintenance of biodiversity, recreation, etc. Domestic water quality is typically defined by taste, odor, color, and organic and inorganic substances posing human health risks (Table 2.2 In the United States, WQ guidelines are either legally enforceable primary standards (Table 2.3) or non-enforceable guidelines (Table 2.4). Water with more than 1000 mg/l of total dissolved solids (TDS) is salty to drink (Table 2.5).

Chemical Properties - Water has ability to act as both an acid and a base and be a solvent for cations, anions and some types of organic matter. Water has high viscosity, cohesion and adhesion, surface tension, melting and boiling points, and the large temperature range through which it is a liquid.

As water travels across the landscape (Fig. 2.2), it interacts with its environment through a variety of chemical processes (Table 2.6). It picks up and transports dissolved gases, cations and anions, amorphous organics, trace metals, and particulates. The most common cations include: calcium (Ca), magnesium (Mg), sodium (Na), potassium (K) and ammonium (NH_4). The most common anions include: nitrate (No_3), sulfate (S), chloride (CL) and different forms of phosphorus (P). Most amorphous substances are organic and carbon based compounds having high cation exchange capacity. Common particulates include: mineral particles, organic debris, and microscopic organisms (plankton, diatoms, etc.). The chemical behavior (Table 2.7) and the origin of contamination (Table 2.8) vary with the type of chemical constituents.

Dissolved gasses – The most abundant dissolved gases in water are nitrogen, oxygen, carbon dioxide (Co_2), methane (Ch_4), hydrogen sulfide (H_2S), and nitrous oxide (N_20). The first three are abundant in the earth atmosphere. The second three are typically products of biogeochemical processes that occur in non-aerated, 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 affect color, taste, odor, and the chemistry of water. Unpolluted surface waters have high DO because of reaeration and oxygen produced by photosynthesis by aquatic plants. Groundwater tends toward oxygen depletion and reducing conditions, because oxygen consumed during hydrochemical and biochemical reactions is not replenished by the atmosphere. Polluted surface waters tend to have lower DO concentrations, because oxygen is consumed during decomposition of organic matter. Super-saturation of DO can occur at the bottom of reservoir spill ways and can apparently cause lethal gas bubble trauma in fish and aquatic life (Bistal and Ruff 1996).

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 subject to re-aeration and mixing by wind action and typically has abundant DO. In contrast, the deepest, stagnant waters contain little or no oxygen, do to oxidation of organic matter. The breakdown of organic matter results in deep waters being acidic and rich in carbonic acid. Consequently, stagnate deep waters have the chemical conditions necessary to produce soluble forms of Fe, Mn, and S, and other taste and odor-producing substances, thus municipal water is usually drawn from the upper oxic surface waters. However, during climatic periods when the temperature of the surface waters rapidly falls and becomes denser than bottom waters, the entire water column of the lake mixes or “turns-over”. During turn-over, the DO content of the entire lake can decrease, cause massive fish kills, and foul smelling and poor tasting water. Similar problems can occur in stratified lakes from mixing by short and intense runoff events.

DO concentrations strongly influences the solubility and stability of elements, including iron (Fe), manganese (Mn), nitrogen, sulfur, and arsenic. When Fe and Mn are exposed to air, they form insoluble precipitates, make water turbid, cause stains in laundry, and impart a bitter taste (Cox 1964). In low DO waters, Fe minerals are reduced, and can release adsorbed phosphorus and other elements. The solubility of most arsenic and arsenic-sulfur compounds are DO dependent, but most forms have concentrations above the primary standard of 0.05 mg/l (Table 2.3) (Freeze and Cherry 1979).

Radon^{222} (Rn^{222}) is a gas derived from the radioactive decay of uranium and thorium and is relatively common in groundwater from rocks and soils that contain radioactive minerals.

Organic compounds - Organic compounds have carbon and usually hydrogen and oxygen, and relatively low solubility’s in water. Their degradation usually involves microorganisms and the chemical processes of hydrolysis, oxidation-reduction and volatilization. In natural waters, they are transported as dissolved phases and attached to particulates. Common, natural organic compounds include plant and animal tissue, and products of their decomposition. Common, synthetic organic compounds include petroleum products and most pesticides and herbicides (Table 2.3). Most synthetic toxic organic compounds originate with coal mining, petrol refining, and from textile, wood pulp and pesticide factories (Table 2.8). Human health affects are related to immediate short-term toxicity or reactions from long-term exposure. Evidence suggests that disinfecting some types organic-rich water with chlorine might lead to the formation of volatile halogenated organic compounds, suppected carcinogens (Gems 1991).

Trace metals and non-metals - The most common water quality criteria for trace metals for drinking water are silver, cadmium, chromium, copper, mercury, iron, manganese, lead, and zinc (Table 2.3 and Table 2.4). Common trace non-metals include carbon, chlorine, sulfur, nitrogen, fluorine, arsenic, selenium, phosphorus and boron. Most of these elements occur in natural, uncontaminated waters in concentrations below 1 mg/l.

Most metals have relatively low solubility's in water. Their solubilities are usually lowest at neutral pH's and increase with increasing acidity and increasing alkalinity. A characteristic feature of metals is their tendency to form hydrolyzed species and/or complex with inorganic anions and organic complexes. These complexes typically absorb to suspended particulates or form insoluble precipitates. Therefore, their transport across the landscape is closely related to sediment transport and the presence of organic compounds, especially fulvic acids (Schlesinger 1997).

Trace metals are natural occurring elements. However, their concentrations in soil and water can be significantly increased over background levels by mining, processing of metal ores, and leaching from domestic and industrial dumps. Some metals, like copper and cadmium, are associated with automobiles and are concentrated on roads and parking lots (Bannerman et al. 1993). Major sources of lead include urban soil and lead-based paint (Mielke 1999). Water-soluble lead acetate is an ingredient in some hair-coloring cosmetics and can contaminate water when spilled in a sink or used in a bathroom. Acid rain or runoff can increase the concentration of metals in both ground and surface waters.

Fluoride is a trace non-metal in nearly all natural waters. For dental health, it is added to some domestic water. Arsenic can be naturally present in waters from areas of recent volcanism and is widely used in pigments, insecticides, and herbicides (Freeze and Cherry 1979). In the ranges of DO and pH of typical groundwater, many arsenic forms are soluble and have concentrations above standards. Selenium is a toxic nonmetallic element, and can occur in appreciable concentrations in shale, coal, and uranium ore (Schlesinger 1997).

Nitrogen - Nitrogen has a dominant role in many basic biochemical reactions. In certain chemical forms, it can adversely affect humans and ecosystems. During industrial times, fertilizer production and other human activities have more than doubled the global rate of nitrogen fixation (Kinzig and Scolow 1994). Thus, nitrogen is the most common water pollutant in the United States. In the Northeastern United States, human activities have apparently increased the nitrate concentrations in major rivers three to tenfold since the early 1900s (Matson et al. 1997).

Certain nitrogen compounds can have toxic effects at relatively low concentrations, for example the "blue baby syndrome" associated with nitrite (NO_2) in drinking water (Table 2.3). The presence of nitrate (NO_3) in water can present health hazards, but the relationships are less clear since humans ingest nitrates in many foods (Gems 1991). Apparently, certain bacteria residing in vertebrate digestive tracts can convert relatively benign nitrate into the toxic nitrite (Kinzig and Scolow 1994). Ammonia can be toxic to some aquatic invertebrates and fish depending on the concentrations of DO, temperature, pH, salinity, and the carbon dioxide – carbonic acid equilibrium of water. Excessive nitrogen concentrations in water can lead to algal blooms, excessive growth of aquatic plants, and eutrophication. Anthropogentic alteration of N cycles can increase acid rain (Vitousek et al. 1997).

In the simplest form, the nitrogen cycle consists of three major reservoirs: the atmosphere, the ocean, and terrestrial ecosystems (Fig 3). However, the flow of nitrogen between reservoirs can occur in many forms and pathways (Fig. 4). Inorganic nitrogen can be transported in water as dissolved nitrogen gas (N_2), ammonia and ammonium cations (NH_3, NH_4) or as anions of nitrite (NO_2) or nitrate (No_3). Their concentrations are low in most unpolluted fresh waters 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/l and 0.1 mg/l (Gems 1991). Nitrate concentrations greater than 1 mg/l generally indicate anthropogentic inputs. Generally,

the lowest concentrations are found in deep ground water and surface waters draining pristine wildlands (Gems 1991, Spahr and Wynn 1997).
The thin, organic-rich material on the forest floor is an important source of organic and inorganic acids produced during the biogeochemical breakdown of organic matter. In general, immobilization of nutrients predominates in the upper layers of fresh forest litter, while mineralization of N, P, and S is usually greatest in the lower forest floor and upper mineral soil (Schlesinger 1997). In undisturbed forest, decomposition and reactions in the upper few centimeters of litter and soil can consume much or all of the infiltrating DO and generate large amounts of organic acids. This decomposition is generally so efficient that only a small proportion of the organic matter produced by a forest every year is added to the long-term storage of soil organic matter or humus. The organic acids and other products produced can cause undesirable odor or taste in water and can increase the calcium and hardness in water draining limestone (Freeze and Cheery 1979).

Organic nitrogen is converted to inorganic nitrogen in an oxidation sequence: ammonium to nitrite to nitrate. In strongly oxidized water, nitrate is the stable phase and is very mobile. As redox potential declines, nitrate is reduced (i.e. denitrified) to N_20 or N_2. Because of the potential adverse ecosystem and health effects associated with nitrites and nitrates, denitrification is desirable from a water quality point of view. Generally, the amount of net mineralization is directly related to the total content of organic nitrogen and the availability of available carbon (C) (Vitousek and Melillo 1979, Schlesinger 1997). Nitrification tends to be lower in soils with low pH, soil oxygen, soil moisture, temperature, or high litter carbon/nitrogen ratios. At the watershed scale, denitrification rates can vary with landscape positions (Peterjohn and Correll 1984, Jordan et al. 1993, McDowell et al. 1992). Generally, relatively high rates are found in riparian forests and at the base of hillslopes where water, C, N and P concentrate.

Seasonal differences in nitrogen uptake by plants can cause measurable variations in concentrations in soil or surface waters, usually lowest during the early growing season when uptake is greatest (Boyd 1996). Maximum nitrate concentrations are typical in the winter months when plant uptake is reduced. However, seasonal trends can be reversed with large anthropogentic inputs of N.

Phosphorus – Phosphorus (P) in drinking water is not considered a human health hazard. However, it's presence can be indicate of organic pollution. Because phosphorus can accelerate the growth of algae and aquatic vegetation, it contributes to eutrophication and associated deterioration of municipal water supplies.

In natural waters, P exists in both dissolved and particulate fractions. Dissolved phases typically originate from excretions by organisms. Particulates phosphorus can have organic or inorganic origins. In streams, a large fraction of phosphorus is absorbed and transported with organic and inorganic particulates. In lakes, a large proportion of the P in oxygenated surface waters is held in plankton biomass (Schlesinger 1997). In deeper anoxic lake waters, P is absorbed to sediments and particulates, but can be released during the reduction of iron compounds. Nearly all P in terrestrial ecosystems is originates from weathering of minerals and the largest flux of P is by rivers (Fig. 5).

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 (Stumm and Morgan, 1970, Freeze and Cherry 1979). This evolution increases with travel time and distance, and it varies between surface and groundwater. Deep groundwater evolution typically involves increases in dissolved solids and decreases in DO, organic waste, pesticides, phosphorus, and nitrogen. In contrast, concentrations of organic waste, pesticides, phosphorus, and nitrogen typically increase as surface waters travel across the landscape and interact with natural and anthropogentic systems.

Fresh, young waters have limited contact with their surroundings. They are generally low in TDS and rich in bicarbonate anions derived from soil Co_2 and the dissolution of carbonate minerals. Sulfate anions tend to dominate in intermediate age groundwater, while chloride anions dominant in older, deep groundwater that has travel long distances. These sulfate and chloride anions are derived from soluble sedimentary minerals. Because these minerals are usually present in small amounts in most rocks, water must travel considerable distances before it is dominated by either sulfate or chloride anions.

The DO content and redox potential tends to decrease as water travels across the landscape. Rain and snow have relatively high concentrations of DO and high redox potentials resulting from exposure to atmospheric oxygen. In organic rich soil layers, oxidation of organic matter removes DO and reduces the redox potential of soil water. As water travels through the subsurface, DO is consumed by aerobic bacteria oxidizing organic matter. Eventually, anaerobic conditions prevail and ammonia, manganese, ferrous iron, and sulfate become major oxidizing agents.

Cation concentrations in water can vary considerable in space and time and do not follow well-defined, theoretically based sequences like anions or redox potentials. Nevertheless, cation concentration typically increases with travel distance in both surface and groundwater. In general, cations enter the aquatic system from the weathering of minerals and the breakdown of organic materials. Calcium and magnesium are the most abundant cations in water supplies and are often removed by chemical treatments to prevent scaling and to reduce the amount of soap needed.

Physical Properties - The physical characteristics of water typically managed for in municipal watersheds are: temperature, color, turbidity, total sediment load, taste and odor.

Temperature - Temperature influences physical, chemical and biological properties of water and is critical to the quality of both raw and treated municipal water. Chemical and metabolic reaction rates, viscosity, and solubility of water are temperature dependent. Aquatic organisms require specific temperature conditions, because metabolism, reproduction, and other physiological processes are controlled by heat sensitive proteins and enzymes (Ward 1985). A 10°C increase in temperature roughly doubles the metabolic rate of cold-blooded organisms and the rate of many chemical reactions (MacDonald et al. 1991). A permanent change in water temperature of 5°C can significantly alter the structure and composition of an aquatic population (Nathanson 1986). Temperature increases decrease DO concentrations, but increases the rate of oxidation and the efficiency of certain biological wastewater-treatment systems.

Water temperature varies naturally with time of day, season, and the type of water body (rainfall, stream flow, groundwater etc.). Surface water temperature respond to seasonal changes in net radiation, daily changes in air temperature, and local variations in incoming radiation. Because of the lack of radiation inputs and the thermal capacity of the subsurface, temperature variation in groundwater are less than for surface water. Anthropogentic activities can increase the temperature of water bodies. Land use can indirectly influences water temperatures by affecting energy inputs, evaporative cooling, and the way water flows across the landscape. For example, removal of forest canopy over small streams increases the range of water temperature. Once warm, the rate surface water cools depends on heat transfer to the atmosphere. Because the thermal energy of streams is not rapidly lost, increases in stream temperatures caused by natural or anthropogentic processes are generally additive and water temperature increases in a downstream direction (MacDonald et al. 1991). Seasonal and spatial temperature variations within reservoirs can have large effects on the quality of raw municipal water (Cox 1964).

Color and turbidity - Pure water is a colorless transparent liquid thin layers and bluish-green in thick layers. However, the presence of sediment, particulate matter, and insoluble inorganic and organic compounds typically add color and reduce transparency and light transmission. Light dependent aquatic organisms, taste, esthetic appeal, and the effectiveness of certain waste treatment processes can be affected.

Turbidity affects water clarity and the scattering of light. Turbidity is typically controlled by suspended particles (Table 2.1). In clear water, approximately 50% of incident light is scattered or transformed into heat within the first meter of water. Turbidity reduces the depth of sunlight penetration, which can alter water temperature and stratification, the rate of photosynthesis of aquatic organisms, and the DO content of the water body. Turbid waters can contain soil particles or fecal matter harboring microorganisms and/or absorbed contaminants. The removal of particulates by gravity and/or chemical additions is typically the first step in treating water for human consumption. The sedimentation of particles and the bleaching action of sunlight on colloidal material during reservoir storage can reduce color and turbidity of water (Cox 1964).

Sediment - Sediment is a major water-quality concern, because it can transport harmful substances, and it impacts water treatment costs and the maintenance of municipal water-distribution systems. While sediment comes from natural weathering and sculpturing of the landscape, accelerated levels of erosion and sedimentation are associated with many anthropogentic activities (Table 2.8).

The general term “sediment” includes both organic and inorganic particles from the physical and chemical erosion of the landscape. Sediment is transported by wind, gravity, and/or water. In water, sediment is transported as suspended load or along the substrate as bed load. Sediment load is the total of suspended plus bed load. 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 in terms of mass per unit area (tons/ac/yr.), concentrations (ppm), or in turns of the lowering the landscape (in./yr.). In general, high sediment loads increase water treatment costs, and reduce the storage volume and life span of water storage facilities.

Biological properties - Water organisms are usually grouped by: (1) those that obtain carbon needed for biosynthesis from carbon dioxide (autotrops), or (2) those using existing organic compounds as their carbon source (heterotrops). Generally, autotrops increase DO concentrations in water through photosynthesis, while heterotrops breakdown and recycle dead organic materials, and decrease DO concentrations.

Most microbial contaminants are caused by heterotrops transmitted to a water system via human and animal fecal matter (U.S. EPA 1999d). Most water-borne pathogenic microorganisms are bacteria or viruses in sewage and septic leachate (Table 2.8). Bacterial pathogens come from both animal and human sources, while viral pathogens are usually of human origins. Viruses infecting animals normally do not cause human disease. However, some animal viruses are suspected to affect humans, particularly those infecting the respiratory system (e.g. sin nombre, hantavirus, influenza, and Ebola viruses). Common bacterial diseases spread by aquatic microorganisms include Legionnaire's disease, cholera, typhoid, and gastroenteritis. Some aquatic-borne viral diseases include polio, hepatitis, and forms of gastroenteritis. Water borne parasitic diseases include amoebic dysentery, flukes, and giardiasis.

Several of these diseases are found in the United States. The analytical procedures for detecting them in water are costly and time consuming. Therefore, most drinking and recreational waters are routinely tested for microbes easy to detect, but are highly correlated with human health hazards. Coliforms are a common type of microbe test (Table 2.1). Their presence and abundance in raw water is used to screen for fresh fecal contamination (Cox 1964), to determine treatment plant efficiency, and to assess the integrity of the distribution system.

Many environmental factors can affect the transport of microbes across the landscape (Table 2.9 and Table 2.10). Relatively coarse-grained or sandy soils are poor adsorbers of microbes (Keswick and Gerba, 1980, U.S. EPA 1999d). In contrast, fine-textured clay soils or soils with abundant colloidal organic material are very adsorbent of microbes. Clay soils slow migration, but can enhance the survival of microbes (Bitton et al. 1986; Keswick and Gerba, 1980).

The pH and ionic strength of liquids percolating past adsorbed microbes can influence virus sorption and desorption. Apparently, weak ionic strength of pore water weakens the virus-soil adsorption forces and increases their detachment and concentrations in percolating fluid (Bitton et al. 1986). Therefore, rainwater with its extremely low ionic strength may mobilize and transporting viruses from the upper layers of the soil.

The absorption of viruses to organic soils is generally poor, as is the adsorption in the presence of high concentrations of dissolved organic matter (Keswick and Gerba, 1980). Humic and fulvic acids are common in forest litter layers and soil, and may cause a loss of virus infectivity and prevent adsorption.

Fecal contamination of surface and ground water can occur by several pathways (Table 2.8). The concentration of microbes in surface runoff is generally higher in warmer months and higher in runoff from grazed rather than ungrazed lands (Edwards et al. 1997). Other important sources of fecal coliforms are domestic animal from lawns and residential streets (Bannerman et al. 1993), leaking sewer lines, substandard well casings, and failed septic systems (U.S. EPA 1999d). The water distribution systems may be contaminated from effluent reaching wells and ground water pressure forcing contaminated ground water into cracked pipes.

Water stored in reservoirs can positively effect microbial content of surface waters, from sedimentation of particles with adsorbed microbes and the germicidal action of sunlight (Cox 1964). However, these effects can vary spatially and seasonally, and by the morphology and chemistry of the water body.

In Lake Okeechobee Florida, microbial concentrations and water quality problems increase during algal blooms (James and Havens 1996). The occurrence of blooms is related to total nitrogen and phosphorus inputs, wind velocity and mixing of the water column. The relative importance of these variables varies in different parts of the lake and at different times of the year.

Hydrologic Cycles and Water Budgets

The annual circulation of water is the largest movement of a chemical substance at the surface of the earth (Schlesinger 1997). The hydrological cycle describes the constant exchange between the land, sea and atmosphere. A water budget is the balance of inflows, outflows and changes in storage at a specific location, typically a watershed (Fig. 2). Both water cycles and budgets considered the flows of water in solid, liquid, or gaseous phase between cycle compartments. Typically, they are viewed in a time sequence from precipitation to stream flow. All nutrients are water soluble to some extent, the hydrologic cycle has an important influence on nutrient cycling. Precipitation brings both water and nutrients into ecosystems, while stream flow is a major pathway of natural nutrient removal. Soil moisture and groundwater are instrumental in weathering, leaching, transport, and plant uptake. Water is involved in the assimilation, consumption, decomposition, and release of nutrients by plants.

The basic equation that describes hydrologic budget can be described as:

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

- Q = stream flow
- P = precipitation
- I = interception
- T = transpiration
- E = evaporation
- G = groundwater
- W = water withdrawals for consumptive use
- R = return flow from outside sources
- S = change in storage over measurement period

Units in the equation are expressed in volumes or lengths per unit time (i.e. million gallons per day - MGD, inches/year – in/yr). The magnitudes of the components are dependent on the spatial and temporal scale considered and their evaluation involves errors due to measurement and interpretation (Winter 1981). Water budgets for small watersheds typically have combined measurements errors of 20% or greater.

Precipitation - Water enters a watershed in many forms. Precipitation is often classified by: it's physical form (liquid, solid, gas), size (rain, drizzle, mist), the major factors responsible for the lifting the air mass to cool and to produce the precipitation (cyclonic, frontal, warm-front, cold-front, convection, or orographic, etc,), and chemistry (acidity). General, annual precipitation increases with elevation and aspect relative to the prevailing winds.

Because the water molecule attracts other molecules, precipitation contains dissolved gases in proportion to their concentrations in the atmosphere, their solubility, and ambient temperature (Blatt et al. 1980). Uncontaminated precipitation has low concentrations of solutes, is slightly to moderately acidic, and has a high redox potential. Except in areas with significant ocean spray, the dominant ion in precipitation is bicarbonate, which is produced by the reaction of carbon dioxide with water. The equilibrium pH for non-saline water in contact with atmospheric Co_2 is 5.7. Rainwater and melted snow in non-urban, non-industrial 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 have a pH as low as 3-4. This "acid rain" is typically a result of NO_3 and SO_4 derived from the incorporation of gaseous pollutants in raindrops (Schlesinger 1997). Increased acidity can increase the rate of weathering of soil minerals and increase the release of cations from exchange sites. Consequently, the concentration of metals in source water and the corrosion of water storage and distribution systems can also increases and result in higher metal concentrations in drinking water (McDonald 1985, Park 1987)

Evaporation, transpiration and evapotranspiration - Evaporation is the process of converting water from a liquid or solid state to a gaseous state. Evaporation from snow and ice, but is most common from lakes and wetlands, soil surfaces, and accumulations of water on vegetation or other surfaces. Evaporation rates depend primarily on solar radiation, temperature, wind, and the humidity gradient above the evaporating surface. Evaporation losses can be large from reservoirs (Viessman, et. al. 1977).

Transpiration is the process by which water is evaporated from air spaces within leaves, is influenced by available soil moisture, type of vegetation, vapor pressure gradients across leaf surfaces, and the factors that influence evaporation. In many cases, evaporation and transpiration are summed and reported as one termed, evapotranspiration (ET). Unlike other pathways, ET recycles water without uptake essential nutrients, thus increasing the concentration of solutes in the remaining water.

Interception, throughfall, and stemflow - Interception is precipitation that adheres to leaves and vegetation, and evaporates instead of falling directly or indirectly to the ground. Throughfall is water that eventually falls from the vegetation to the ground, while 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, structure and composition of the canopy, season, and the form of precipitation (Anderson et al. 1976, Scatena 1990). In general, forests intercept more than grasslands and conifers intercept more water than hardwoods.

Throughfall and stemflow can represent both an input and portion of nutrient cycling within the forest system. Nutrient inputs result from the washing of accumulated atmospheric deposition of nutrient and particulates from vegetative surfaces. The recycled component represents leaching of nutrients from plant tissue. In general, the cation and anion concentrations in throughfall are 2 to 100 times those of rainfall (Wenger 1994). In deciduous forest, the highest concentration of nutrients in throughfall and stemflow occurs during the summer months when the forest has the largest leaf area

Soil water - Once precipitation or throughfall reaches the forest litter layer and soil surface, it either infiltrates, evaporates, temporarily ponds, or leaves the area as surface runoff. Infiltrated water can reside in many subsurface areas (Fig. 6) and remain subsurface for periods ranging from seconds to many centuries. Infiltration rates are influenced by the magnitude and intensity of rainfall, the type, density and extent of vegetal cover, and the temperature and condition of the soil surface.

Groundwater - As water percolates into deep groundwater, TDS, calcium bicarbonate, magnesium bicarbonate, calcium sulfate and magnesium sulfate concentrations generally increase. In contrast, concentrations of DO, organic wastes, pesticides, phosphorus, and nitrogen tend to decrease. Thus, older and deeper groundwater generally has better water quality and needs less treatment than surface water or shallow, young groundwater.

The purification and evolution of groundwater occurs because of chemical absorption on the surfaces of soil particles and microbial decomposition. Microbial processes are usually responsible for methanogenesis, denitrification, Fe and SO_4 reduction, and the breakdown of synthetic organic compounds. Denitrifying bacteria have been observed at depths of 300 meters, while sulfate-reducing bacteria can remove most SO_4 within 15m of the soil surface (Schlesinger 1997). The travel time and distance needed to remove viruses or synthetic organics can vary considerable but typically occurs on the order of days to years and meters to kilometers (Table 2.9).

Stream flow – Stream flow is commonly divided into two types: stormflow and baseflow. Water is transported to streams by three basic processes: overland flow or surface runoff, interflow or subsurface stormflow, and groundwater flow (Linsley et al. 1982). Interflow is water that infiltrates into the upper soil layers and moves laterally until it enters a stream channel. In most undisturbed forests, infiltrate rates exceed rainfall intensities. Thus, overland flow is limited to a disturbed areas or to areas with shallow, degraded soils. In contrast, interflow is common, especially in areas with thin, porous soils. Interflow leads to quick saturation of soils at the base of slopes, which discharge into stream channels. The stream network expands to the headwaters during rainfall events to carry this discharged water. After the rainfall ends, interflow decreases, the area of saturated soils decrease, and stream flow decreases resulting in the stream network to contract. This process is called the variable source concept.

The chemical characteristics of stream water depend on its source, flow path, and transit time to the stream. In general, the concentrations of dissolved solids decrease with increasing discharge, increase with the length of the flow path from precipitation to deep groundwater, and the amount of time traveled across or through the landscape. This is especially true of highly soluble and non-biologically limiting ions associated with chemical weathering like Ca, Mg, Na, Si, Cl, SO_4, and HCO_3 (Schlesinger 1997). In contrast, sediment and particulate matter from erosion tend to increase with stream discharge and the proportion of discharge from surface runoff.

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

As streams move from headwaters to lowlands their morphology, water chemistry and biotic communities change. In general, headwater streams are shaded by terrestrial vegetation and have biotic communities that are dependent on leaf litter and terrestrial sources of organic matter (Vannote et al. 1980). These streams tend to have relatively young water and lower concentration of TDS. At lower elevations, the amount of light entering water, the contribution of groundwater, and anthropogentic contaminants generally increases. Consequently, TDS and contaminants tend to increase and aquatic plants and algae become the major source of organic matter. As dissolved constituents are transported downstream, they are converted to organic forms, accumulated in organisms and 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 et al. 1982).

The physical and biotic changes in water quality occurring along the river can affect the operation and cost of municipal water treatment. Because TDS and pollution concentrations increase as water progresses downstream, water withdrawn from lower elevations streams typically needs more treatment than water from undisturbed, forested headwaters. Nevertheless, because of greater volumes and less seasonal variable in water supply, lowland intakes can be more reliable sources of water.

Water withdrawals and return flow - Water withdrawal (W) removes water from a hydrologic system and conveys it to a place for a consumptive use (Table 2.11). Non-consumptive uses or instream uses include navigation and recreation. Return flows (R) is water added to a stream after being withdrawn from source within or outside the watershed and used. Return flows are typically from leaky pipes or irrigation ditches, irrigation, sewage plants, or industrial sources.

The stream flow required to dilute and assimilate municipal and industrial return flow depends on the quality and quantity of the return flow, on the quality of the receiving waters, and on the length and turbulence of flow in the receiving waters (Gupta 1995). The quality of return flow from sewage plant reflects the quality of wastewater entering the plant and the type of treatment. In secondary, sewage treatment facilities 80 to 90% of the organic matter is typically removed (Hunter and Kotalik 1973). Unless the sewage effluent is disinfected, most wastewater treatments do not markedly reduce pathogens. In the United States, most effluent is disinfected by chlorination. However, excessive chlorination without dechlorination may lead to toxicity problems for aquatic organisms.

Drinking Water Treatment

Most raw water is not suitable for human consumption without treatment. Some water only needs to be filtered and disinfected before consumption (National Academy 1997). Other waters must be treated with additional processes to remove specific chemical contaminants or nuisance chemicals like iron and manganese. Tables 2.12 to 2.15 present information on water treatment techniques that can be used for controlling common contaminants. While the table provides guidance on selecting the appropriate treatment processes, the selection of the best treatment process must be made on a site-specific basis by a water treatment specialist. Additional information can be found in recent volumes of *Water Quality and Treatment*, published by the American Water Works Association, Denver, Colorado and *Safe Water from Every Tap*, National Academy Press, Washington D.C., 1997.

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Table 2.1: Chemical units and definitions commonly used in water quality analysis. Modified from U.S. EPA 1999a and other sources

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 (e.g., pathogens).

Anions: Negatively charged ions. The most common in natural waters are nitrate (No_3), sulfate (S), chloride (CL) and different forms of phosphorus (P).

Biological Oxygen Demand, BOD: mg/l of dissolved oxygen required to decompose biodegradable organic material.

Cations: Positively charged ions. The most common in natural waters are calcium (Ca), magnesium (Mg), sodium (Na), potassium (K) and ammonium (NH_4).

Chemical Oxygen Demand, COD; mg/l of dissolved oxygen required to decompose biodegradable and non-degradable organic material.

Chronic Health Effect: The possible result of exposure over many years to a drinking water contaminant at levels above its MCL.

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.

Community Water System: A water system which supplies drinking water to 25 or more of the same people year-round in their residences.

Compliance: The act of meeting all state and federal drinking water regulations.

Contaminant: Anything found in water (including microorganisms, minerals, chemicals, radionuclides, etc.) which may be harmful to human health.

Cryptosporidium: 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.

Disinfectant: A chemical (commonly chlorine, chloramine, or ozone) or physical process (e.g., ultraviolet light) that kills microorganisms such as bacteria, viruses, and protozoa.

Distribution System: A network of pipes leading from a treatment plant to customers' plumbing systems.

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, Hepatitis E virus, Norwalk viruses, astrovirus and caliciviruses.

Equivalents per liter; the number of moles of solute multiplied by the valence of the solute species in 1 liter of solution.

Eutrophication; enrichment of waters with nutrients, especially phosphorus and nitrogen that leads to enhanced plant growth, algal blooms and depleted oxygen levels as this plant material decays.

Finished Water: Water that has been treated and is ready to be delivered to customers.

Giardia lamblia: 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 systems pump and treat from aquifers

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.

Mass concentration: The mass of a solute dissolved in a specific unit volume of solution, usually expressed in mg/l.

Maximum Contaminant Level (MCL): The highest level of a contaminant that EPA 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.

Molality: the number of moles of solute in a 1 kg mass of solution.

Molarity: the number of moles in a 1m3 of solution.

Monitoring: Testing that water systems must perform to detect and measure contaminants. A water system that does not follow EPA's monitoring methodology or schedule is in violation, and may be subject to legal action.

Non-Transient, Non-Community Water System: A water system which supplies water to 25 or more of the same people at least six months per year in places other than their residences. Some examples are schools, factories, office buildings, and hospitals which have their own water systems.

Organic Contaminants: Carbon-based chemicals, such as solvents and pesticides, which can get into water through runoff from cropland or discharge from factories. EPA has set legal limits on 56 organic contaminants.

Pathogen: A disease-causing organism.

PH: A common measure of acidity defined as the negative logarithm of the hydrogen ion concentration. The pH scale runs from 0 to 14. Acid solutions have a pH less than 7.0 and alkaline solutions have a pH greater than 7.0. Pure distilled water has a neutral pH of 7.0, whereas vinegar and soap have approximate pH values of 3.0 and 10 respectively.

Public Water System (PWS): Any water system which provides water to at least 25 people for at least 60 days annually.

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.

Secondary Drinking Water Standards: Non-enforceable federal guidelines regarding cosmetic effects (such as tooth or skin discoloration) or aesthetic effects (such as taste, odor, or color) of drinking water.

Sole Source Aquifer: An aquifer that supplies 50 percent or more of the drinking water of an area.

Source Water: Water in its natural state, prior to any treatment for drinking.

Surface Water: The water that systems pump and treat from sources open to the atmosphere, such as rivers, lakes, and reservoirs.

Transient, Non-Community 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 which 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.

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 mg/l.

Turbidity: The cloudy appearance of water caused by the presence of tiny particles. One turbidity unit is the interference in the passage of light caused by a suspension of 1 mg/l of silica. Turbidity equal to or less than 5 TU is generally not noticeable to the average person.

Vulnerability Assessment: An evaluation of drinking water source quality and its vulnerability to contamination by pathogens and toxic chemicals.

Watershed: The land area from which water drains into a stream, river, or reservoir.

Wellhead Protection Area: The area surrounding a drinking water well or well field which is protected to prevent contamination of the well(s).

Table 2.2: Common types of water guidelines for different water uses. Adapted after GEMS 1991

	Human Consumption	Irrigation	Livestock	Fisheries	Recreation
Coliform bacteria	*				*
Nematode eggs		*			
Particulate matter	*			*	
Dissolved Oxygen, (i.e. BOD, COD)				*	*
Nitrates	*	*	*		
Nitrites	*		*	*	
Salinity	*	*	*	*	
Inorganic pollutants (i.e. trace metals)	*	*	*	*	*
Organic pollutants	*			*	*
Pesticides	*			*	

Table 2.3. National Primary Drinking Water Regulations, from U.S. EPA 1999b. Where MCLG = Maximum Contaminant Level Goals, MCL = Maximum Contaminant Level (MCL) or the maximum permissible level of a contaminant in water which is delivered to any user of a public water system. TT = Treatment Technique.

Contaminants	MCLG (mg/L)	MCL or TT (mg/L)	Potential Health Effects from Ingestion of Water	Sources of Contaminant in Drinking Water
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	0.05	Skin damage; circulatory system problems; increased risk of cancer	Discharge from semiconductor manufacturing; petroleum refining; wood preservatives; animal feed additives; herbicides; erosion of natural deposits
Asbestos (fiber >10 micrometers)	7 million fibers per Liter	7 MFL	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	0.004	0.004	Intestinal lesions	Discharge from metal refineries and coal-burning factories; discharge from electrical, aerospace, and defense industries
Cadmium	0.005	0.005	Kidney damage	Corrosion of galvanized pipes; erosion of natural deposits; discharge from metal refineries; runoff from waste batteries and paints
Chromium (total)	0.1	0.1	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=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)	0.2	0.2	Nerve damage or thyroid problems	Discharge from steel/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	Action Level=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	0.002	0.002	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 six months - life threatening without immediate medical attention.	Runoff from fertilizer use; leaching from septic tanks, sewage; erosion of natural deposits
Nitrite (measured as Nitrogen)	1	1	"Blue baby syndrome" in infants under six months - 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	0.0005	0.002	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	TT	Nervous system or blood problems; increased risk of cancer	Added to water during sewage/wastewater treatment
Alachlor	zero	0.002	Eye, liver, kidney or spleen problems; anemia; increased risk of cancer	Runoff from herbicide used on row crops
Atrazine	0.003	0.003	Cardiovascular system problems; reproductive difficulties	Runoff from herbicide used on row crops
Benzene	zero	0.005	Anemia; decrease in blood platelets; increased risk of cancer	Discharge from factories; leaching from gas storage tanks and landfills
Benzo(a)pyrene	zero	0.0002	Reproductive difficulties; increased risk of cancer	Leaching from linings of water storage tanks and distribution lines
Carbofuran	0.04	0.04	Problems with blood or nervous system; reproductive difficulties.	Leaching of soil fumigant used on rice and alfalfa
Carbon tetrachloride	zero	.005	Liver problems; increased risk of cancer	Discharge from chemical plants and other industrial activities

Chlordane	zero	0.002	Liver or nervous system problems; increased risk of cancer	Residue of banned termiticide
Chlorobenzene	0.1	0.1	Liver or kidney problems	Discharger from chemical and agricultural chemical factories
2,4-D	0.07	0.07	Kidney, liver, or adrenal gland problems	Runoff from herbicide used on row crops
Dalapon	0.2	0.2	Minor kidney changes	Runoff from herbicide used on rights of way
1,2-Dibromo-3-chloropropane (DBCP)	zero	0.0002	Reproductive difficulties; increased risk of cancer	Runoff/leaching from soil fumigant used on soybeans, cotton, pineapples, and orchards
o-Dichlorobenzene	0.6	0.6	Liver, kidney, or circulatory system problems	Discharge from industrial chemical factories
p-Dichlorobenzene	0.075	0.075	Anemia; liver, kidney or spleen damage; changes in blood	Discharge from industrial chemical factories
1,2-Dichloroethane	zero	0.005	Increased risk of cancer	Discharge from industrial chemical factories
1-1-Dichloroethylene	0.007	0.007	Liver problems	Discharge from industrial chemical factories
cis-1, 2-Dichloroethylene	0.07	0.07	Liver problems	Discharge from industrial chemical factories
trans-1,2-Dichloroethylene	0.1	0.1	Liver problems	Discharge from industrial chemical factories
Dichloromethane	zero	0.005	Liver problems; increased risk of cancer	Discharge from pharmaceutical and chemical factories
1-2-Dichloropropane	zero	0.005	Increased risk of cancer	Discharge from industrial chemical factories
Di(2-ethylhexyl)adipate	0.4	0.4	General toxic effects or reproductive difficulties	Leaching from PVC plumbing systems; discharge from chemical factories
Di(2-ethylhexyl)phthalate	zero	0.006	Reproductive difficulties; liver problems; increased risk of cancer	Discharge from rubber and chemical factories
Dinoseb	0.007	0.007	Reproductive difficulties	Runoff from herbicide used on soybeans and vegetables
Dioxin (2,3,7,8-TCDD)	zero	0.00000003	Reproductive difficulties; increased risk of cancer	Emissions from waste incineration and other combustion; discharge from chemical factories
Diquat	0.02	0.02	Cataracts	Runoff from herbicide use
Endothall	0.1	0.1	Stomach and intestinal problems	Runoff from herbicide use
Endrin	0.002	0.002	Nervous system effects	Residue of banned insecticide
Epichlorohydrin	zero	TT	Stomach problems;	Discharge from industrial

			reproductive difficulties; increased risk of cancer	chemical factories; added to water during treatment process
Ethylbenzene	0.7	0.7	Liver or kidney problems	Discharge from petroleum refineries
Ethelyne dibromide	zero	0.00005	Stomach problems; reproductive difficulties; increased risk of cancer	Discharge from petroleum refineries
Glyphosate	0.7	0.7	Kidney problems; reproductive difficulties	Runoff from herbicide use
Heptachlor	zero	0.0004	Liver damage; increased risk of cancer	Residue of banned termiticide
Heptachlor epoxide	zero	0.0002	Liver damage; increased risk of cancer	Breakdown of hepatachlor
Hexachlorobenzene	zero	0.001	Liver or kidney problems; reproductive difficulties; increased risk of cancer	Discharge from metal refineries and agricultural chemical factories
Hexachlorocyclopentadiene	0.05	0.05	Kidney or stomach problems	Discharge from chemical factories
Lindane	0.0002	0.0002	Liver or kidney problems	Runoff/leaching from insecticide used on cattle, lumber, gardens
Methoxychlor	0.04	0.04	Reproductive difficulties	Runoff/leaching from insecticide used on fruits, vegetables, alfalfa, livestock
Oxamyl (Vydate)	0.2	0.2	Slight nervous system effects	Runoff/leaching from insecticide used on apples, potatoes, and tomatoes
Polychlorinated biphenyls (PCBs)	zero	0.0005	Skin changes; thymus gland problems; immune difficiencies; reproductive or nervous system difficulties; increased risk of cancer	Runoff from landfills; discharge of waste chemicals
Pentachlorophenol	zero	0.001	Liver or kidney problems; increased risk of cancer	Discharge from wood preserving factories
Picloram	0.5	0.5	Liver problems	Herbicide runoff
Simazine	0.004	0.004	Problems with blood	Herbicide runoff
Styrene	0.1	0.1	Liver, kidney, and circulatory problems	Discharge from rubber and plastic factories; leaching from landfills
Tetrachloroethylene	zero	0.005	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 (TTHMs)	none[5]	0.10	Liver, kidney or central nervous system problems; increased risk of cancer	Byproduct of drinking water disinfection
Toxaphene	zero	0.003	Kidney, liver, or thyroid problems; increased	Runoff/leaching from insecticide used on cotton

			risk of cancer	and cattle
2,4,5-TP (Silvex)	0.05	0.05	Liver problems	Residue of banned herbicide
1,2,4-Trichlorobenzene	0.07	0.07	Changes in adrenal glands	Discharge from textile finishing factories
1,1,1-Trichloroethane	0.20	0.2	Liver, nervous system, or circulatory problems	Discharge from metal degreasing sites and other factories
1,1,2-Trichloroethane	0.003	0.005	Liver, kidney, or immune system problems	Discharge from industrial chemical factories
Trichloroethylene	zero	0.005	Liver problems; increased risk of cancer	Discharge from petroleum refineries
Vinyl chloride	zero	0.002	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
Radionuclides				
Beta particles and photon emitters	none[5]	4 millirems per year	Increased risk of cancer	Decay of natural and man-made deposits
Gross alpha particle activity	none[5]	15 picocuries per Liter (pCi/L)	Increased risk of cancer	Erosion of natural deposits
Radium 226 and Radium 228 (combined)	none[5]	5 pCi/L	Increased risk of cancer	Erosion of natural deposits
Microorganisms				
Giardia lamblia	zero	TT	Giardiasis, a gastroenteric disease	Human and animal fecal waste
Heterotrophic plate count	N/A	TT	HPC has no health effects, but can indicate how effective treatment is at controlling microorganisms.	n/a
Legionella	zero	TT	Legionnaire's Disease, commonly known as pneumonia	Found naturally in water; multiplies in heating systems
Total Coliforms (including fecal coliform and *E. Coli*)	zero	5.0%	Used as an indicator that other potentially harmful bacteria may be present	Human and animal fecal waste
Turbidity	N/A	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
Viruses (enteric)	zero	TT	Gastroenteric disease	Human and animal fecal waste

Table 2.4. National Secondary Drinking Water Regulations which are non-enforceable guidelines regulating contaminants that may cause cosmetic effects (such as skin or tooth discoloration) or aesthetic effects (such as taste, odor, or color) in drinking water. From U.S. EPA 1999c

Contaminant	**Secondary Standard**
Aluminum	0.05 to 0.2 mg/L
Chloride	250 mg/L
Color	15 (color units)
Copper	1.0 mg/L
Fluoride	2.0 mg/L
Foaming Agents	0.5 mg/L
Iron	0.3 mg/L
Manganese	0.05 mg/L
Odor	3 threshold odor number
pH	6.5-8.5
Silver	0.10 mg/L
Sulfate	250 mg/L
Total Dissolved Solids	500 mg/L
Zinc	5 mg/L

Table 2.5: A Total Dissolves Solid classification of water and major and minor constituents in groundwater. Adapted from Freeze and Cherry 1979.

Category	Total Dissolved Solids, mg/l
Fresh water	0-1000
Brackish water	1000-10,000
Saline water	10,000-100,000
Brine water	> 100,000
Major Constituents	Typically > 5 mg/l
Bicarbonate (HCO_3)	
Calcium (Ca)	
Sodium (Na)	
Cloride (Cl)	
Sulfate (SO_4)	
Magnesium (Mg)	
Carbonic acid (HCO_2)	
Minor Constituents	Typically (0.01-10 mg/l)
Ammonium (NH_4)	
Boron (B)	
Carbon Dioxide (CO_2)	
Carbonate	
Fluoride (F)	
Iron (Fe)	
Manganese (Mn)	
Nitrate (NO_3)	
Oxygen (O_2)	
Phosphorus (P)	
Potassium (K)	
Strontium (Sr)	

Table 2.6: Common chemical processes involved as water interacts with its environment.

Acid-base reactions are a common type of chemical reactions 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 substance that containing 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). Acidic soil or waters can have increased concentrations of metals and decrease 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, is the exchange of chemicals from solution and the surfaces of particles by chemical or physical bonding. When the adsorption bonds are chemical they are relatively irreversible whereas if they are physical (i.e. van der Waals 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 absorption and subsequent settling of sediments important process in lakes, river and water treatment plants.

Volatilization, or 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 surface by wind or heat. Particular important process in burned areas (see Fire Chapter) and after the application of pesticides or nutrients to soils (see pesticide Chapter).

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 which are both 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. The solubility of Oxygen in water is relatively low (9 mg/l at 25C), and oxygen is used in consumed by hydrochemical and biochemical reactions it must be replenished

Oxidation-reduction By definition, 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 electrodes 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 organic form. This breakdown usually involves soil microbes and is caused by come combination of photolysis, hydrolysis, oxidation-reduction, and enzyme actions. Immobilization is the accumulation of N,P and other nutrients in soil microbes.

Table 2.7. Summary of the chemical behavior of important water contaminants. Adapted and updated from Foster and Gomes, 1989. Where P = reactions probable occur, M = reactions may occur, and D = reactions do occur

CONTAMINANT	BIOCHEMICAL TRANSFORMATIONS		CHEMICAL REACTIONS		PHYSIOCHEMICAL RETARDATION	
	Aerobic	Anaerobic	Acid	Alkaline	Acid	Alkaline
Metals						
Aluminium (Al)	M	P	P	D	P	D
Cadmium (Cd)	P	P	P	D	P	M
Chromium (Cr)	P	P	M	P	D	P
Copper (Cu)	P	P	M	D	D	M
Iron (Fe)	D	P	P	D	M	D
Lead (Pb)	P	P	M	D	M	D
Manganese (Mn)	M	M	P	D	M	D
Mercury (Hg)	M	P	M	D	M	D
Silver (Ag)	P	P	M	D	M	
Zinc (Zn)	P	P	P	D	P	D
Inorganic Non-metals						
Ammonium (NH_4)	D	P	P	P	P	D
Nitrate (NO_3)	P	D	P	P	P	P
Sodium (Na)	P	P	P	P	P	M
Sulphate (SO_4)	P	D	P	M	P	P
Fluoride (F)	P	P	M	M	D	P
Chloride (Cl)	P	P	P	P	P	P
Arsenic (As)	P	P	M	P	M	D
Selenium (Se)	P	P	D	M	D	P
Cyanide (Cn)	P	P	P	P	D	P
Organic Compounds						
Aliphatic Hydrocarbons	D	P	P	P	D	D
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	
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

Table 2.8. Common waterborne pathogenic and indicator bacteria and viruses. After U.S. EPA 1999d.

Waterborne Pathogenic Bacteria	Waterborne Pathogenic Viruses
Legionella Mycobacterium avium intracellular (MAC) Shigella (several strains) Heliobacter pylori Vibrio cholerae (V. cholerae) Salmonella typhi (S. typhi) Salmonella typhimurum (S. typhimurum) Yersinia Campylobacter (several strains) Escherichia coli (E. coli) (several 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 Escherichia coli (E. coli) Enterococci Fecal Streptococci Clostridium perfringens (anaerobic spores) Klebsiella pneumoniae Aeromonas hydrophila	Bacteriophage Bacteroides phage Coliphage Male-specific coliphage FRNA phage FDNA phage Host Salmonella 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

Table 2.9 Factors influencing virus transport and fate in the subsurface. After U.S. EPA, 1999d

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	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;	Unknown	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	Virus adsorption to particle surfaces and hence their survival increases in unsaturated conditions	Increased saturation promotes desorption of viruses from particle surfaces and migration in ground water

.

Table 2.10: Summary of activities generating subsurface contamination. Adapted and updated from Foster and Gomes, 1989. Where N = nutrient, FC = fecal coliform, O = organic load, S = synthetic organic compounds, HC = Hydrocarbons, DOC = dissolved organic carbon, H = heavy metals, I = inorganic salts

Activity	Spatial Distribution	Major types of pollution	Pollution Indicators
Urbanization			
Unsewered Sanitation	Point, Diffuse	N,F,O,S	NO_3, NH_4, FC, DOC, Cl
Leaking sewers	Point, Line	N,F,O,S	FC, NH4, NO_3
Leaking fuel tanks	Point	O	HC, DOC
Highway 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, NO_3, HC
Agriculture			
Cropland	Diffuse	N,O,S,P	NO_3, sediment
Livestock	Point, Diffuse	F,N,O	NO_3, sediment
Mineral Extraction	Point, Diffuse	H,I	Variable, sediment

Table 2.11: Definitions of water use. Adapted from Solley et al. 1998.

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 US in 1995 was estimated at 14% of withdrawals and deliveries.

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.

Conveyance loss: The quantity of water that is lost in transit from its source to point of use or point of return.

Domestic use: Includes water used for normal household purposes, such as drinking, food preparation, bathing, washing cloths and dishes, flushing toilets, and watering lawns and gardens. The consumptive use of water for domestic purposes in the US in 1995 was estimated at 26% of withdrawals and deliveries.

Industrial use: Includes water use for processing, washing, and cooling in facilities that manufacture products like steel chemical, paper and petroleum refining. The consumptive use of water for industrial purposes in the US in 1995 was estimated at 15% of total withdrawals and deliveries.

Instream use: Water use that takes place without the water being diverted or withdrawn from surface or ground water sources. Examples include hydroelectric power generation, navigation, freshwater dilution of saline estuaries, maintenance of minimum stream flows to support fish and wildlife habitat, and waste water 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 US in 1995, 19% was lost in conveyance, 61% was consumptive use, and 20% was returned to surface of ground-water supplies.

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 US in 1995 was estimated at 26% of withdrawals and deliveries.

Mining use: Offstream 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 US in 1995 was estimated at 27% of withdrawals and deliveries.

Offstream use: Water that is diverted or withdrawn from a surface or groundwater source and conveyed to the place of use.

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.

Return flow: The quantity of water that is discharged to a surface or ground-water source after release from the point of use and thus becomes available for further uses.

Thermoelectric power: Includes offstream uses for the generation of electric power with fossil-fuels, nuclear, or geothermal power. In the US in 1995, surface water supplied more that 99% of the thermoelectric withdrawals. Consumptive use was about 2% of withdrawals.

Wastewater Release: Includes the disposal of water conveyed through a sewer system. In the US in 1995 approximately 2% of these releases were reclaimed for beneficial uses such as irrigation of golf courses and public parks.

Withdrawal: The quantity of water diverted or withdrawn from a surface or ground-water source.

Water Delivery: The quantity of water delivered to a specific point of use

Water Release: The quantity of water released after a specific use

Table 2.12. Water Treatment Technologies by type of disinfectants and oxidants. This table is adapted from *Safe Water From Every Tap*, "Chapter 3," pages 50-57, published by National Research Council, Washington, DC, 1997.

	Free Chlorine	Chloramine	Chlorine Dioxide	Ozone	Potassium Permanganate	Ultraviolet Radiation	Aeration
General Water Quality Parameters							
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							
VOCs							X
Semivolatile compounds							
Pesticides/herbicides							
Biodegradable organic matter							
Inorganic Chemicals							
Hardness							
Iron	X		X	X	X		X
Manganese	X		X	X	X		
Arsenic							
Selenium							
Thallium							
Fluoride							
Radon							X
Radium							
Uranium							
Cations							
Anions							
Total dissolved solids							
Nitrate							
Ammonia							

Table 2.13. Water Treatment Technologies by type of adsorption and ion exchange system. This table is adapted from *Safe Water From Every Tap*, "Chapter 3," pages 50-57, published by National Research Council, Washington, DC, 1997.

	Powdered Activated Carbon	Granular Activated Carbon	Ion Exchange	Activated Alumina
General Water Quality Parameters				
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				
VOCs	X	X		
Semivolatile compounds	X	X		
Pesticides/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				

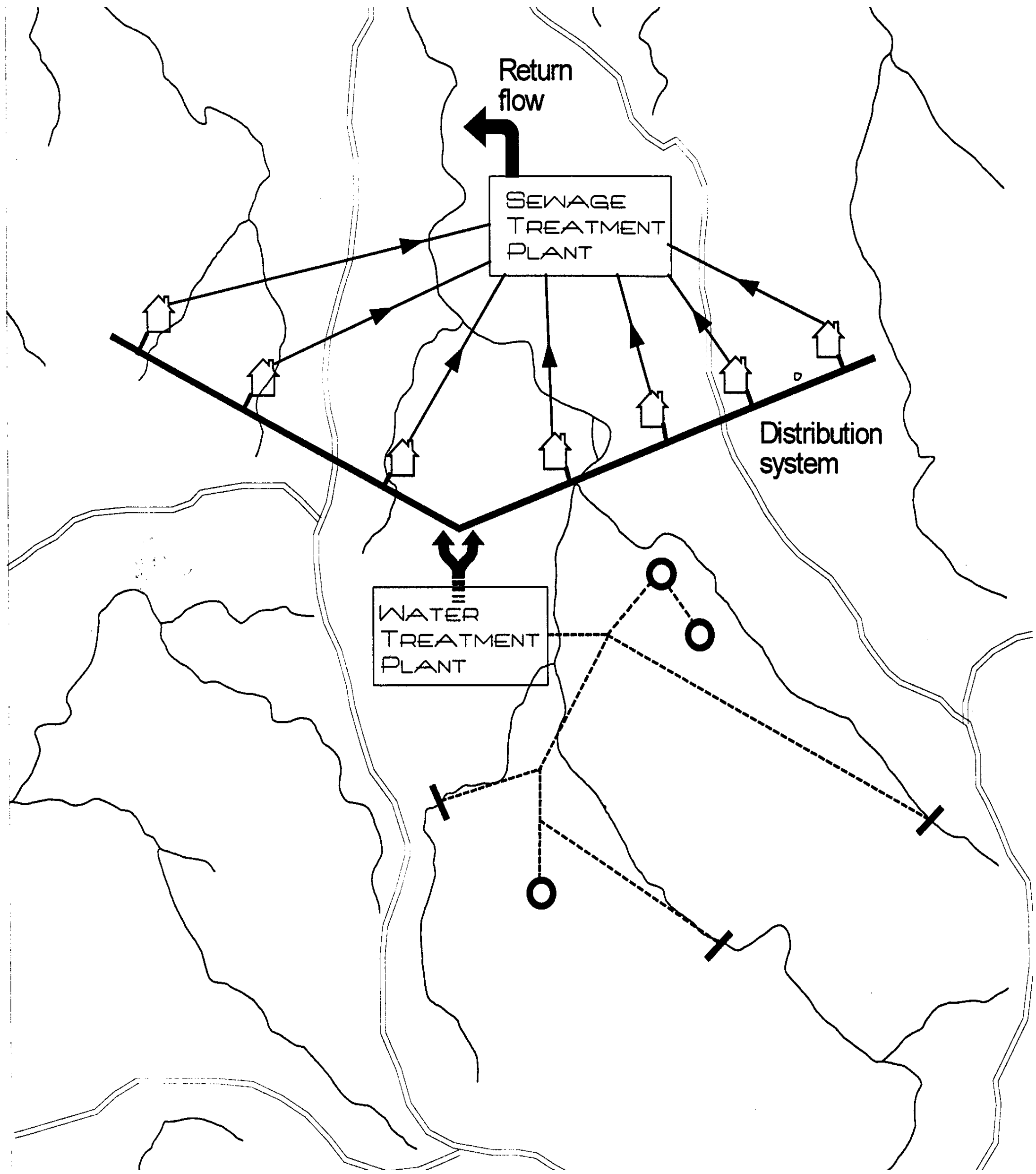
Table 2.14. Water Treatment Technologies by membrane processes. This table is adapted from *Safe Water From Every Tap*, “Chapter 3,” pages 50-57, published by National Research Council, Washington, DC, 1997.

	Microfiltration	Ultrafiltration	Nanofiltration	Reverse Osmosis	Electrodialysis/ ED Reversal
General Water Quality Parameters					
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					
VOCs					
Semivolatile compounds				X	
Pesticides/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					

Table 2.15. Water Treatment Technologies by type of filtration system. This table is adapted from *Safe Water From Every Tap*, “Chapter 3,” pages 50-57, published by National Research Council, Washington, DC, 1997. **X*** when operated in a biologically active mode.

	Direct Filtration	Conventional Filtration	Dissolved Air Flotation	Precoat Filtration	Slow Sand Filtration	Bag/Cartridge Filters	Lime Softening
General Water Quality Parameters							
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							
VOCs							
Semivolatile compounds							
Pesticides/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**		

Figure 2.1 Watershed, water treatment and distribution systems.



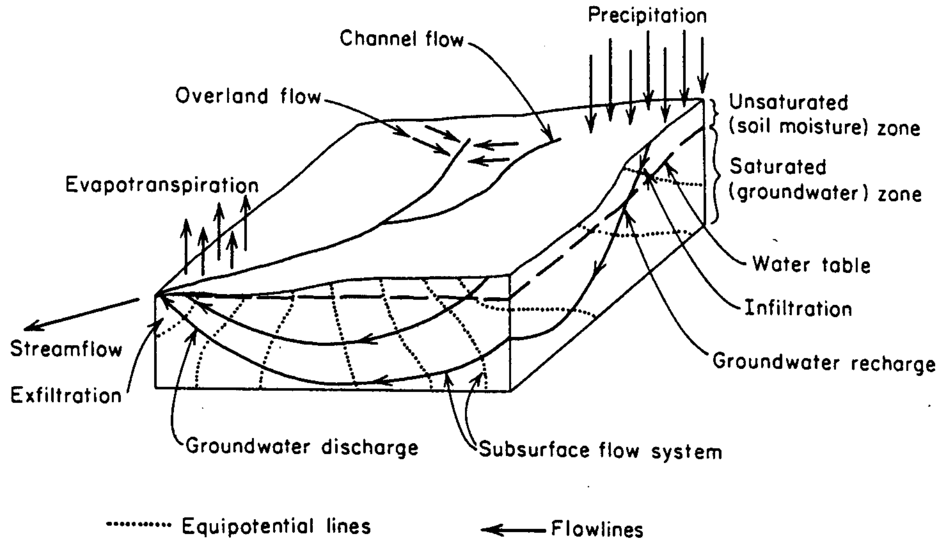


Figure 1.1 Schematic representation of the hydrologic cycle.

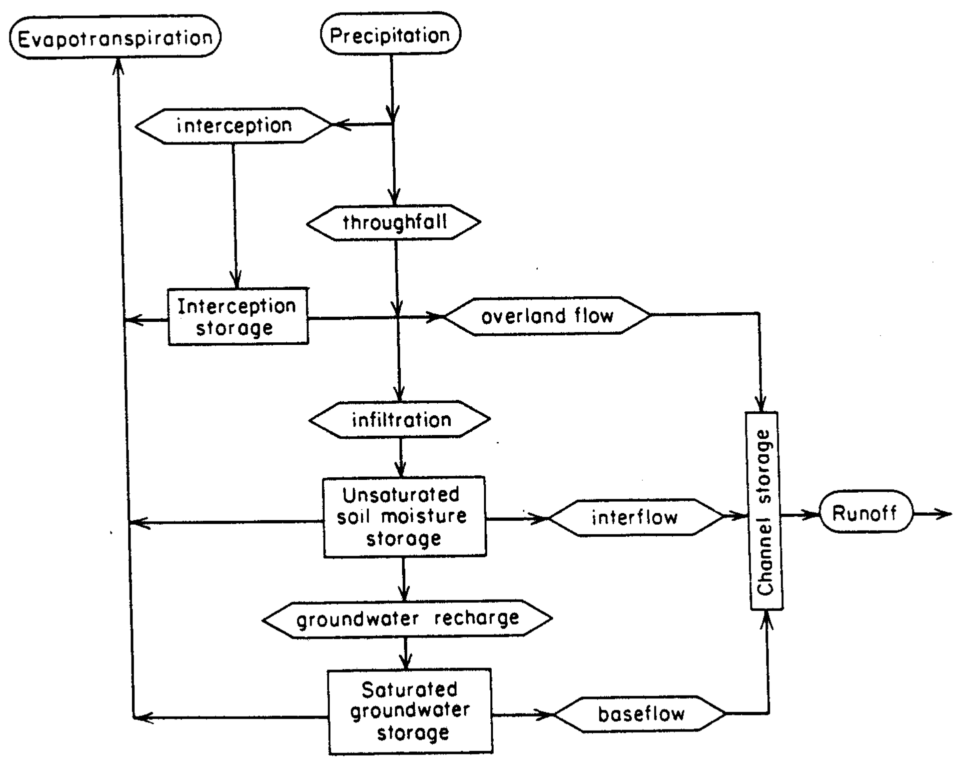


Figure 1.2 Systems representation of the hydrologic cycle.

Figure 2.2. Schematic and system representations of the hydrologic cycle.

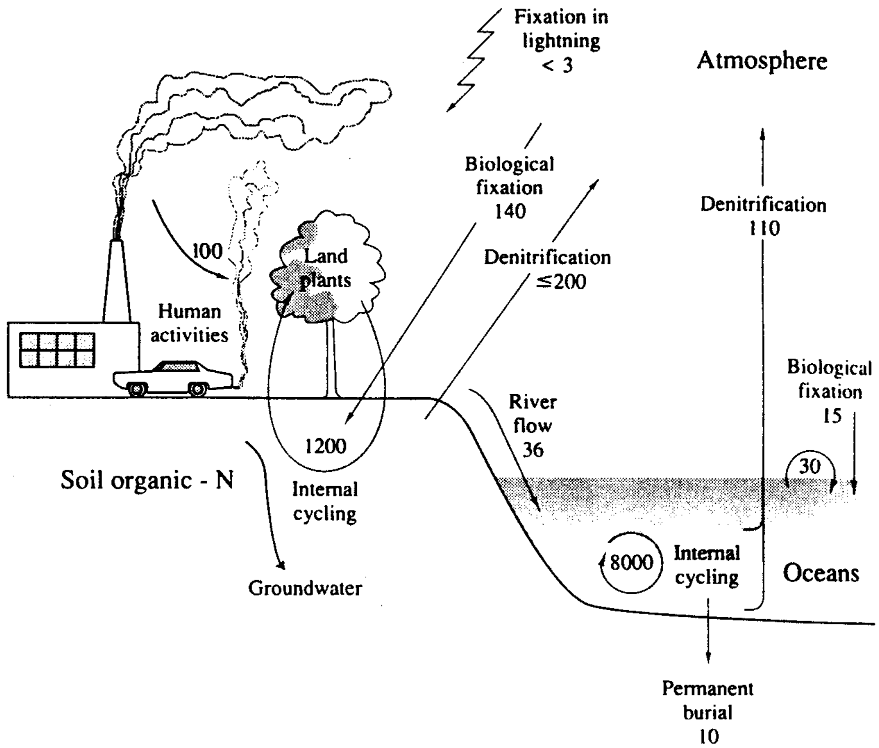


Figure 2.3. Global nitrogen cycle..

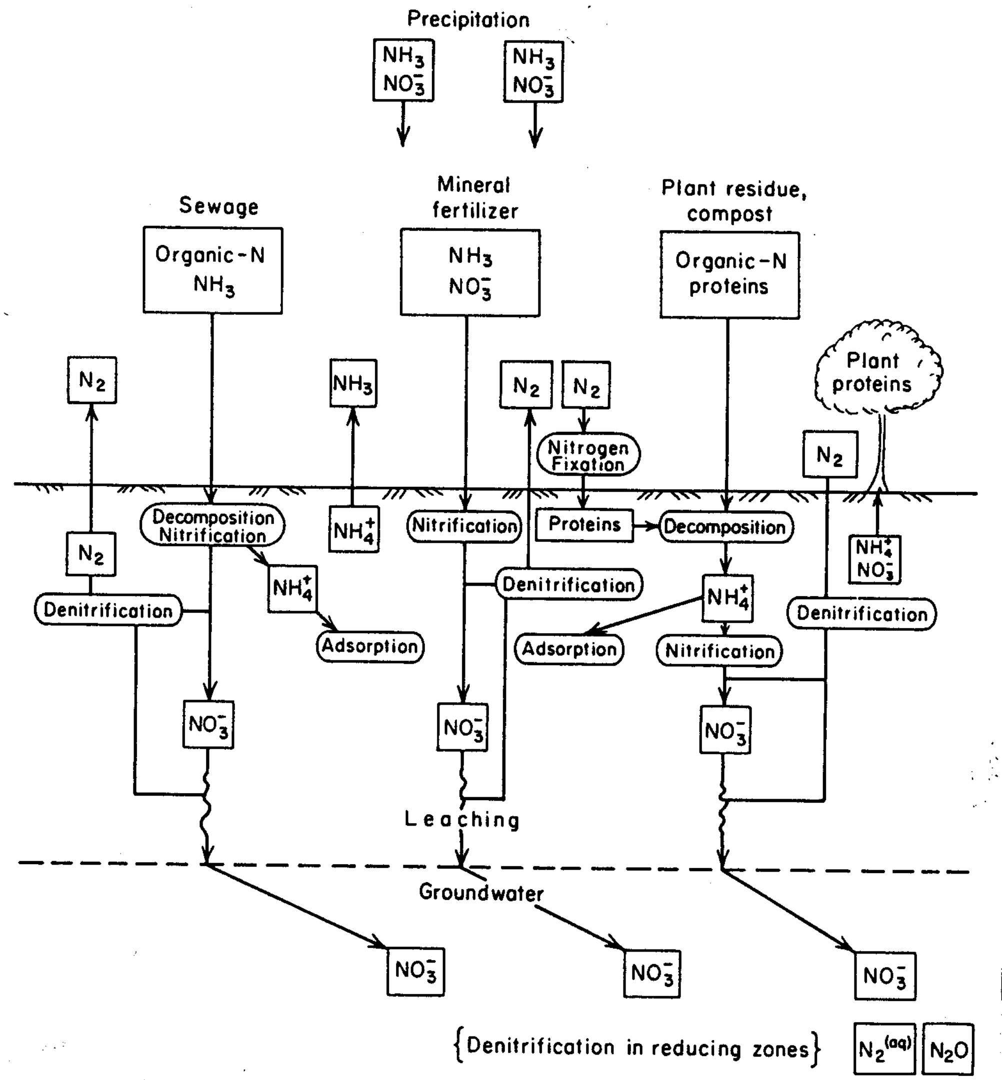


Figure 2.4. Sources and pathways of nitrogen in the subsurface environment (Freeze and Chevy 1979).

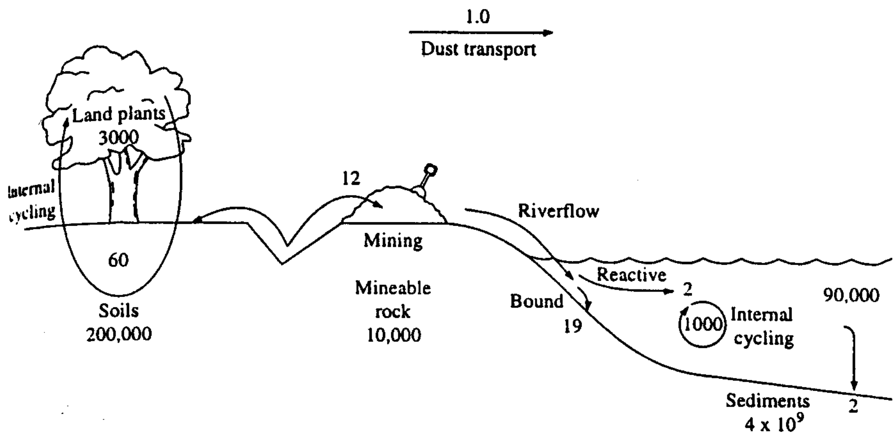


Figure 2.5. The global phosphorus cycle (Schlesinger 1997). Each flux shown in units of 10^{12} g/yr.

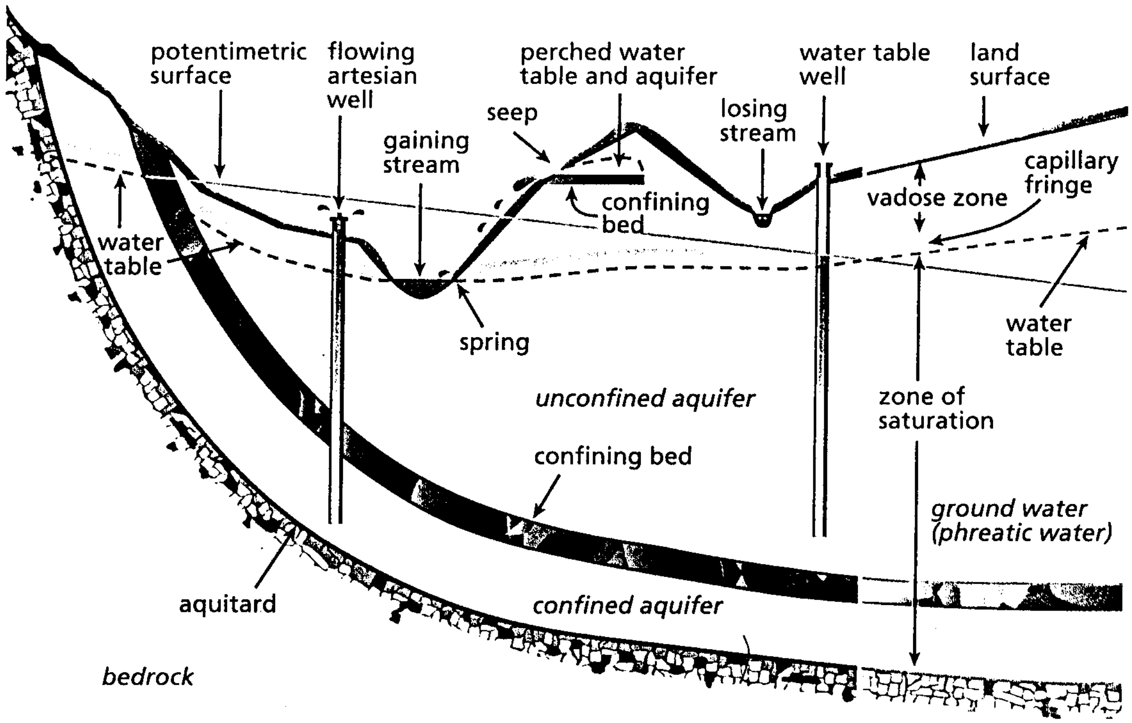


Figure 2.6. Groundwater related features and terminology.

Chapter 3

Watershed Management and Drinking Water: Economic Issues

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Introduction

No other resource serves so many purposes as does water. It is widely used in industry, in electric energy production, in farming and ranching, and of course by households for drinking, washing, and lawn and garden watering. 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 from industrial and sewage plants, commercial establishments, houses, and farms and urban landscapes. Unfortunately, processing wastes often seriously conflicts with the other uses requiring various levels of purity.

Further complicating the picture, water runoff carries sediment from forests, farms, pastures, and other land areas. Often, soil movement is an economic loss to the landowner from which it came, and a problem for the downstream water user. Soil loss decreases site productivity. Sediment impairs fish habitat, accumulates in reservoirs and other water management facilities, and increases water treatment costs. Though some erosion is natural, sediment can be a pollutant. It is greatly exacerbated by some land use activities. In addition, sediment often brings attached pesticides, nutrients, and other contaminants.

Water supply and quality are thus integrally linked. Most water uses are affected by water quality and in turn affect quality for others use. These interdependencies make both water treatment and watershed management essential.

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

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 1970s 98% of the total suspended solids, over 85% of the phosphorus and nitrogen, and 57% of the 5-day BOD in U.S. waters were attributable to nonpoint sources. For 1986, the EPA (1987) reported that nonpoint-source pollution was the cause of 65% of the water quality-impaired stream miles and 76% of the impaired lake acres. The most recent EPA water quality inventory, for 1996, reports a similar finding (EPA 1998). Although agriculture is by far the largest source of water pollution in the U.S., forestry and other activities are important sources in some areas (EPA 1998).

Since the 1972 Clean Water Act was passed, some progress has been made in improving the nation's water quality. Lettenmaier (1991) examined trends at 403 NASQAN stations 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 has remained largely unchanged. In general, successes are associated with point source controls. Little success was found for nonpoint sources. Though considerable investments have been point source control, such findings suggest the nation's water quality goals will not be met without increased control of nonpoint source pollution.

The provision of high quality drinking water is affected by a host of natural events and human activities. The natural events include extreme precipitation events, forest fire, and of pathogens from wild animals. The human activities include mining, agricultural tillage, timber harvest, livestock grazing, automobile use, road construction and maintenance, and use of fertilizers and pesticides. The interactions among these factors, and the unpredictable nature of some, make water quality protection a challenging task.

The costs of water quality control in the U.S. are substantial and rising. In 1985, households obtaining their water from municipal systems spent 0.6% of their income for water, plus an additional 0.4% for wastewater treatment (Singh et al. 1988). These costs were expected to increase by about 30% in response to stricter standards implemented beginning in the late 1980s. A recent EPA survey indicates that community water systems in the U.S. 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.

For water quality, perhaps the most fundamental economic question is whether the benefits of alternative 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 using water for drinking, including human health losses and associated health care costs. Second, by meeting the standard by controlling upstream sources of pollution, benefits include averted losses between the pollution source and the drinking water diversion, such as, fish population, costs of removing sediment from canals and reservoirs, and decreases in recreation quality and use. The costs compared with such benefits include treatment cost at water plants or by rural households, and costs of upstream pollution control. Potential upstream pollution control costs include crop losses from decreased pesticide use[1], erosion control[2], improved road design to limit erosion, in beef production from fencing cattle out of riparian areas, and upstream wastewater treatment.

Despite serious efforts to estimate the benefits of drinking water standards and other water quality controls (Freeman 1982; Freeman 1993), the estimates remain rough. For this reason and for reluctance to compromise drinking water safety, drinking water standards are often set without definitive economic analysis. However, economics still has a primary role in determining how the standards, once set, should be met. It is this topic on which we will focus.

To avoid waste of resources, standards should be met efficiently. A drinking water standard may be met solely by drinking water treatment plants or rural households prior to use, or by a combination of water treatment and pollution control upstream where the water quality problems originate. Since pollution occurs at many points in the watershed, many different costs may be relevant. The costs of water quality protection actions may differ considerably at each source, 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

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[1] For an example of such costs see Easter (1993) on effects of reduced herbicide use on Minnesota farms.

[2] For examples of agricultural erosion control costs, see Lyon and Farrow (1995), Young et al. (1991), and Chang et al. (1994). Binkley and Brown (1993b) summarize studies reporting erosion control costs in forestry.

[3] Another economic issue, related to the equity issue and not discussed here, is the economic impact of pollution in terms of jobs and income. Economic impacts are often of great public interest; indeed, the EPA's "Clean Water Action Plan" (February, 1998) in its section on "economic benefits of clean water" only mentions as benefits the economic impacts of clean water, including those following from angling, commercial fishing, swimming and other recreation activities, and irrigation.

reaching the downstream water quality standard may unfairly allocate costs. If so, options for cost sharing and 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, farm, or 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 treatment to lower of the concentration before it is used (*T*):

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

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

Pollutant concentrations at the reception point are the result of many upstream actions. For example, the city's water treatment plant receives pollution from the forest, the recreation area, the upstream town's sewage plant and storm runoff, rural septic systems, and farms (Fig. 1). In addition, upstream consumptive use (i.e. farms, towns, and trans-basin diversions) can increase the concentration of pollutants reaching the receptor and natural processing of pollutants in the stream or riparian area can decreases concentrations.

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

$$Xr_j = \left(\sum_i e_i \quad {}_{ij} + b_j \right) \Big/ Q_j \tag{3}$$

The streamflow amount is equal to the normal (i.e., virgin) flow (*Qv*) minus upstream consumptive use (*Qu*) resulting from each upstream diversion *k*:

$$Q_j = Qv_j - \sum_k Qu_{kj} \tag{4}$$

The transfer coefficient a varies from 0 to 1 and reflects the water treatment that naturally occurs between the emission and the receptor. For degradable pollutants, this ecological service increases (causing α to decrease) with distance, all else equal. However, for non-degradable pollutants $\alpha = 1$.[5]

The economic task is to determine the most cost-effective way to reach the goal characterized in equation 1. Because pollution can be controlled at its source or removed at the point of

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

[5] If the water body is a lake rather than a stream, the situation is similar except that 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.

reception, and can be lessened by dilution, upstream emitters, upstream consumptive users, and downstream receptors are all candidates for actions to help meet the drinking water standard. Each actor has a cost for reducing the concentration 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 to the receptor (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):

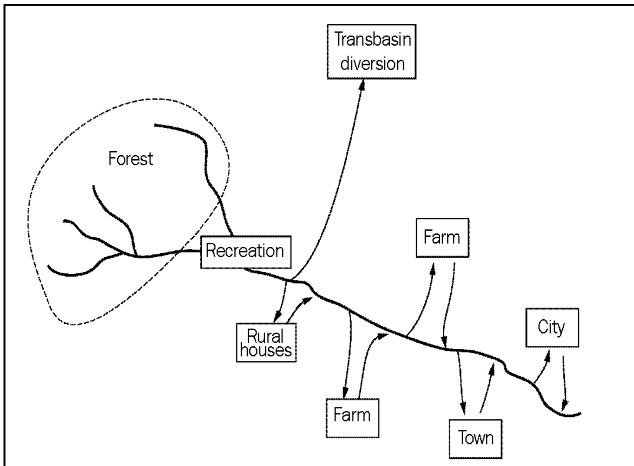

Figure 1
Hypothetical basin

$$\min C_j = \min\left(\sum_i Ce_{ij} + \sum_k Cu_{kj} + Ct_j\right) \tag{5}$$

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, the 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, livestock fences, agricultural irrigation machinery, homeowners' septic systems, and canal sizes are fixed. In the long run, fixed costs are subject to change, such as, 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.

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

2).[7] If no effort is made to control sediment emissions, the concentration of the pollutant reaching a water use reception point is Xe, and the marginal cost of control is $0. Initial reductions in sediment concentration reaching the reception point are likely to be relatively inexpensive, for example, by cleaning culverts and drainage ditches. However, further reductions in sediment concentrations are likely to be progressively more expensive, as indicated by the increasing emitter marginal cost[8] in Figure 2. Reducing the concentration to zero may be quite expensive, and could require closing the road all together.[9]

Now consider the short run marginal cost curve of a downstream water treatment plant. The receptor treatment cost curve (Fig. 2)[10] is likely to rise as the pollutant concentration level is lowered, because ever 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, because of the need to maintain the labor and materials necessary to meet a water quality threat at all times; even if the water quality standard were set at Xe. The receptor still needs to maintain the daily capability of handling water withdrawals with pollutant concentrations that exceeded Xe. Thus, as shown in Figure 2, the receptor's marginal cost curve flattens out to the right but remains above the horizontal axis.

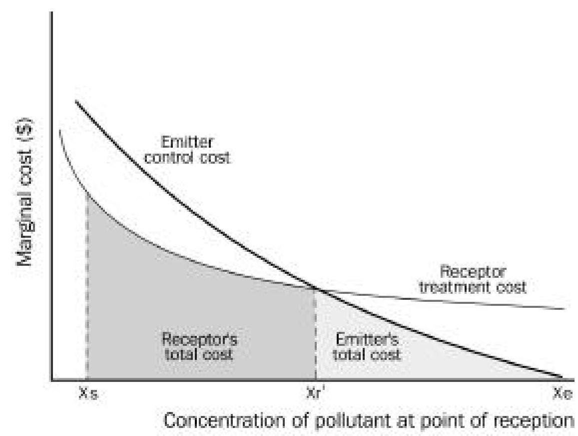

Figure 2
Efficient allocation of cost of meeting drinking water quality standard, with one emitter and one receptor

Assuming a single emitter and single receptor represented by the two short-run marginal cost curves in Figure 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

[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 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, is likely to be similar to that of the emitter shown in Figure 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, Oregon. 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.

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 5, is minimized by finding the equi-marginal 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 2, the emitter's total cost is represented by the area below the emitter's marginal cost curve to the right of Xr', while the area below the receptor's marginal cost curve to the left of Xr' and right of Xs represents the receptor's total cost.

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 equi-marginal 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 et al. (1996, table 3.1) report average costs to reduce a kilogram of phosphorus of $26 with onsite pollution control practices in agriculture versus $169 at municipal treatment plants (1990 dollars)

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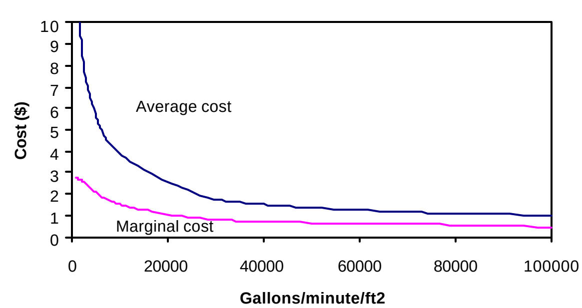
Long Run Costs

Often cost minimization involves long run decisions. Long run cost-curves for water treatment plants depict how costs change as plant capability increases to handle a given pollutant. Most curves have focused on changing water volumes, show economies of scale from considerable decreases in average costs as plant size increases, along with decreasing or relatively constant long run marginal cost curves. For example, Figure 3 shows construction cost for a pressure filtration plant as estimated by Gumerman et al. (1978), expressed in 1978 dollars.[12]

Figure 3. Construction cost of pressure filtration plant, from Gumerman et al. (1973)



When comparing treatment plant costs with costs of controlling pollution at its source, the most relevant issue is not water volume but is concentration. 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 are not inactivated by chlorine at dosages feasible in drinking water treatment. If these oocysts must be removed, new facilities, such as filtration or use of ozone, may 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 of the drinking water provider (Fig. 4). Unlike curves in Figure 3, this long run marginal cost as a function of pollutant concentration is upward sloping or upward stepping. It is such steps that upstream pollution control may help avoid.

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

Complexity

Although straightforward in concept, accurate cost minimization in 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 (NPS), 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. It is even, in the case of sediment and other natural pollutants, often difficult to separate NPS emissions from background levels.

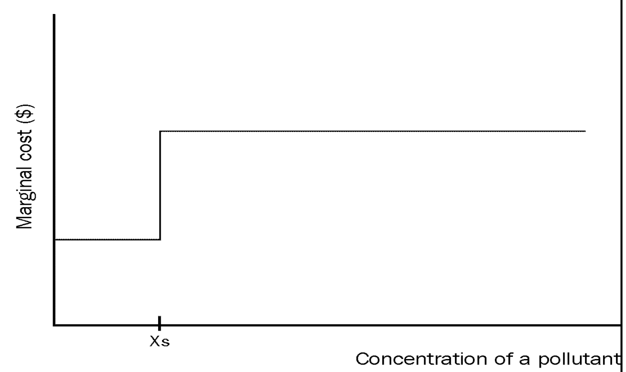


Figure 4
Long run marginal cost as a function of pollutant concentration

Six additional factors further complicate assessment and minimization of the costs of meeting drinking water goals. First, there may be numerous pollution sources and numerous points of consumptive use. Thus, computing the minimum cost for a given receptor may require estimating many different costs. Second, a basin is likely to have numerous drinking water reception points. Third, each emitter and receptor must be concerned with numerous 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. Fourth, each emitter, consumptive user, and diverter may have several options for lowering *Xr*. For example, a forest manager may lower stream sediment loads by careful placement of skid trails, improved construction of roads, and avoiding harvest near streams. Fifth, *Xr* is stochastic, depending on unpredictable and perhaps highly intermittent weather events and uncertain actions of upstream landowners.[13] Finally, other uses beyond drinking water (e.g., fish habitat, recreational swimming, 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 water quality goals across all types of use in a watershed, then efficiency decisions will take account of all water uses, not just drinking water.

The measurement of many of the components of a watershed's cost minimization problem for pollution control is difficult, because of the stochastic nature of nonpoint source pollution. This contributes 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 difficult to know just what to do and where to do it to minimize costs of meeting water quality goals. Therefore in this context, water quality control must 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

[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 expose soil, such as forest fire, heavy 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 et al. (1993) for more on the policy and economics of nonpoint source pollution control in forest areas.

term because nonpoint source pollution depends on weather events with given magnitudes and frequency.

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

Cost Savings from Targeting Upstream Control Efforts

Several studies have estimated the cost savings from replacing so-called command and control strategies of pollution control which emphasize uniform controls with careful targeting of upstream control efforts. Most such studies have dealt with agricultural pollution. Two of the studies, summarized here, have examined schemes that require all emitters or all sub-watersheds with emissions to share proportionally in the control efforts.

An early study (Johnson 1967) examined dissolved oxygen levels in the Delaware River Basin using a model that divided the basin into zones, characterized 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 cost 1.4 to 3.1 times the most inexpensive strategy of carefully targeted control efforts.[14]

A study of reducing phosphorus levels by 50 percent in the Fox-Wolf River basin of Wisconsin (Schleich, White, and Stephenson 1996) compared meeting the target in each of 41 sub-watersheds with meeting the target at the river's mouth in Green Bay. Municipal, construction, and agricultural emissions were modeled. Meeting the goal by sub-watershed 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 were agricultural, are selected for phosphorus reduction. The primary cost savings occur from consideration of loading factor differences among sub-watersheds and not forcing watersheds with already low levels of phosphorus emissions (usually those without major agricultural sources) to participate in the proportional reduction scheme.

Other studies have focused on the command and control strategy of requiring each emitter to reduce pollution loadings 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 the cost savings achievable by using a control strategy that allows differential amounts of control as long as the downstream or ambient goal is reached (Tietenberg 1985, Anderson et al. 1997)

Studies of nonpoint source pollution from agriculture have demonstrated how costs of reaching downstream water quality goals are minimized by carefully selecting pollution control locations and levels. Studies of soil loss from a 1,064 acre watershed in Illinois (Braden et al. 1989) and from a 11,400 acre watershed in Minnesota (Kozloff, Taff, and Wang 1992) found significant cost savings in meeting goals by taking into account the farm-specific costs of reducing emissions and loading factor differences. In the Illinois study, careful targeting reduced roughly 80 the area requiring changes in management. 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 50 percent or more when control efforts were carefully targeted.

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[14] Tietenberg (1985) summarizes two additional BOD studies that giving similar results to the Delaware River study.

Bringing About an Efficient Cost Allocation

Much of the economic writing on pollution (Baumol and Oates 1975; Freeman 1990; Freeman, Haveman, and Kneese 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, where the efficient mechanisms (emission taxes or subsidies, and tradable permits) can work well. These mechanisms have not been easily adapted to the control of nonpoint source pollution.

The principal problem in designing an economic incentive mechanism for nonpoint source pollution control is, as mentioned above, 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 complexity of sediment and pollutant movement and the resultant errors in prediction, have hindered adoption of models to enable the use of economic incentives. 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), it is claimed that education and most other non-regulatory approaches have failed to provide sufficient motivation for major changes (Adler 1992).

Although nonpoint source pollution 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. Such negotiations have been called point/nonpoint source pollution trading, but essentially they are a subsidy scheme (Malik, Larson, and Ribaudo 1994). 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, provide 90% of New York City's water supply. Because of past efforts at watershed protection, a series of city-owned reservoirs that allows long storage times and flexibility in meeting demands, and the low population density in the watersheds, the city has avoided installing filtration for this system (Ashendorff et al. 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 et al. 1997). Instead, the city will invest approximately $1.2 billion over the next few years in a series of efforts to protect the quality of the water entering the city's water treatment plants.[16] Components of this investment include the following:

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[15] New York city is unusual in this sense. Over 90% of surface water systems in the U.S. use filtration (Raucher et al. 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.

- 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 non-city entities, such as inspection and rehabilitation of septic systems, improvements of sewer systems, better storm water management, environmental education, stream corridor protection, and improved storage of sand, salt, and deicing materials.
- paying farmers to follow BMPs
- enhanced monitoring.

In addition, the agreement places restrictions 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.

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.

A problem with such 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, Larson, and Ribaudo 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.

Another benefit of local watershed-based agreements is that they allow for participation of additional parties, including those who would benefit from the agreement but are generally too poorly funded to initiate the process. Because upstream pollution control may benefit not only a downstream city at its water treatment plant, but also water users at various points between the upstream control point and the downstream treatment plant (e.g., fish habitat, reservoir and canal management, and instream recreation). Many parties are potentially interested in the outcome and may be enticed to participate in an agreement initiated by the drinking water provider.[17]

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).[18]

See the following web sites for more details: http://www.state.ny.us/watershed and http://www.epa.gov/region02/water/nycshed.

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

[18] 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; Segerson 1990; Shortle and Abler 1997; Shortle and Dunn 1986).

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.

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Chapter 4

Environmental Setting of Watersheds

The environmental setting of watershed used as sources of drinking water can affect the background levels of water quality constituents. The three environmental aspect are ecosystem influences, atmospheric deposition, and succession.

Ecosystem Influences

Extreme Events Including Floods and Landslides

Fred Swanson

Introduction - Much of the nation's drinking water supply in mountainous regions comes from forested, headwater basins. Many of these watersheds are considered to be in "near-natural" states and, therefore, sources of water with exceptionally high quality. This point concerning naturalness deserves qualification on two counts: (1) natural conditions and processes commonly include substantial variability in streamwater quantity and quality, including periods of poor quality for human consumption, and (2) most seemingly pristine forest landscapes have actually experienced extensive and even intensive land uses now little recognized and partially masked by forest cover. Nevertheless, forested and grassland watersheds have been good, reliable sources of drinking water throughout human history.

As pressure increases to deliver more water of high quality to growing populations, it is important to review the current state of knowledge concerning how natural characteristics of watersheds affect their ability to produce high quality drinking water. In these source areas, drinking water quality can be influenced by natural biological and geophysical features and processes, which can enhance or degrade water quality. Plant species that dominate certain stages of vegetation succession, for example, may be sources of chemical compounds that degrade water color or chemical composition. Certain soil types can produce high levels of turbidity during large flood events, even in watersheds without significant land use history. Some geologic conditions result in undesirable, natural levels of heavy metals and other chemical pollutants. Upland and riparian vegetation and floodplain groundwater systems (hyporheic zones) may remove components of the chemical and suspended sediment load of the streamwaters. Vegetation and soil can buffer effects of atmospheric deposition and suppress production of dissolved and particulate materials in runoff.

Issue and Risks - The general issue considered in this chapter is: How do natural ecological and geophysical processes affect drinking water supplies? Natural processes can enhance or degrade water quality. The specific drinking water quality considerations we wish to address in this context include:

- effects of vegetation change through succession;

- effects of infrequent, extreme events (e.g., floods, droughts, landslides, insect defoliation);

- effects of vegetation in mitigating effects of extreme events and chronic processes;

- effects of vegetation and soil in filtering, buffering, and otherwise mitigating effects of atmospheric deposition (e.g., nitrogen saturation, acid rain).

An understanding of these processes is the basis for science, management, and policy concerning water supplies from several perspectives. First, it is important to have realistic expectations of water quality produced from specific basins. Understanding natural characteristics of watersheds, such as propensity to produce high levels of turbidity during and after major floods, is essential in design of water treatment facilities capable of meeting user needs. Second, assessment of effects of land use practices on water quality and quantity should be set in the context of these background conditions and variability in order to distinguish management effects from natural factors. Third, watershed management and restoration strategies should be founded on understanding of watershed capabilities so objectives can be met with reasonable efficiency. Other chapters in this volume consider effects of land use practices on these natural conditions.

The scope of issues related to ecosystem influences on drinking water quality ranges across a wide variety of water quality factors and aspects of geography. These include, but are not limited to, suspended sediment, turbidity, water color, dissolved constituents, pH, temperature, and dissolved oxygen. There is strong regionality to water quality issues across the United States. Conifer forests and unstable geologic terranes of the west create some conditions distinctive from those of old geologic terrains and deciduous forests of the eastern United States and the subdued topography of grasslands in the mid-continent. Types and degrees of human intervention in these ecosystems exhibits a strong geographic pattern. We principally address forest lands, but these issues also arise in grassland and wetland contexts.

Findings from Studies - Ecosystem properties influence drinking water quantity and quality in many ways, producing high natural variability in flow and water characteristics that have important implications for drinking water use. Influences on water quality may be physical or biological, transient or persistent, localized or widely distributed. Biotic, climatic, and geophysical factors can all strongly alter water quality.

Biotic influences on water quality include both system-level and single species effects. Ecosystem-level influences occur as a result of variation in nutrient uptake during vegetation succession (Swank and Vose 1997). However, Vitousek and Reiners' (1975) hypothesis, developed in part through studies in the Northeast, that old-growth forests are leakier of nutrients than younger forests has not been widely substantiated (e.g., Martin 1979). Individual species, such as red alder (*Alnus rubra*, bong.), a common riparian species in the Pacific Northwest, can alter water color and other characteristics to an extent potentially significant to human consumption (Taylor 1983, Wigington et al. 1998). The extent and water-quality significance of such species typically varies through the course of vegetation succession. "Black water" streams draining organic soils experience persistent poor water-color quality. Biotic disturbance agents, such as insects responsible for extensive defoliation, can alter nitrogen concentration, pH, and other water quality parameters by accelerating nutrient cycling and the acid-neutralizing capacity of ecosystems. This has been well documented in

Appalachian forests (Eshleman et al. 1998) and probably occurs in western conifer forests where intensive and extensive defoliation is also common.

Extreme climatic events commonly initiate changes in water quality by direct effects on vegetation and soil that enhance the availability of dissolved constituents in ground and surface waters. Anomalously cold periods have been observed to lead to high nitrate concentrations in runoff from northeastern deciduous forests (Mitchell et al. 1996). Droughts have been observed to increase nutrient concentrations in both watershed and experimental plot studies (Aber and Driscoll 1997, Xu et al. 1998).

Effects of floods, landslides, and chronic processes on sediment production have been widely studied and are considered most relevant to effects of forest 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 (Fredriksen et al. 1975, Likens and Bormann 1995, Swank and Crossley 1988, Binkley and Brown 1993), landslide inventories (Sidle et al. 1985), sediment budget analyses (Swanson et al. 1982, Reid and Dunne 1996), magnitude-frequency analysis (e.g., Wolman and Miller 1960), as well as studies directly targeting water quality issues for particular storm events (Bates et al. 1998). The latter type of study is most germane to our topic here, but commonly lost 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 runoff of quality that causes problems for use as drinking water is regulated by soil properties, topography, climate, and vegetation conditions. It is widely recognized that steeper slopes favor sediment production. Certain rock and soil types are prone to landslides (Sidle et al. 1985) and production of fine sediment that can cause persistent turbidity (Youngberg et al. 1975). Effects of climate are multifaceted; more precipitation favors water-driven erosion processes, but wetter conditions also favor vegetation development. Vegetation suppresses soil erosion by development of a litter layer protecting the soil from surface erosion and root strength which contributes to soil strength.

Numerous studies in steep, unstable mountain lands have documented that a substantial share of long-term sediment production occurs during extreme events, particularly where landsliding is involved (Swanson et al. 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 et al. 1985). Large, slow-moving landslides, commonly termed earthflows, are also natural, in some cases persisting for millennia in the landscape, and may be more prone to produce persistent turbidity, because the montmorillonite clays that degrade water quality also cause the slow, creeping-style of deformation characteristic of type of landslide (Taskey et al. 1978). Earthflows slowly encroach on stream channels, constricting them over periods of years, before flood waters undercut the toe of the earthflow, causing streamside slides and delivering turbidity-producing sediment to the stream system.

Interactions among geomorphic processes can increase sediment availability that degrades water quality over many years. Very major floods can trigger massive sediment delivery to channels often by landslides, increasing suspended sediment loads in subsequent high flow events in subsequent years. In these cases large amounts of sediment are delivered from hillslopes and incorporated in transient storage sites in river systems, ready to be mobilized in subsequent events of lesser magnitude (Brown and Ritter 1971). Pulses of sediment input to rivers can lead to channel aggradation, widening, and lateral cutting into floodplain deposits and toes of hillslopes, thus entraining sediment that had been stored for long periods. In some cases this stored sediment and colluvium may have undergone sufficient weathering to contain clay minerals that contribute to high levels of turbidity. In this way, a major event can affect more chronic processes operating in inter-flood periods. These relationships are best known from areas of extreme sedimentation in response to natural and/or land use factors (e.g., Kelsey 1980), but these interactions probably operate less conspicuously in systems with less extreme rates of sediment input.

Many of these relations are exemplified by a study of the circumstances leading to the city of Salem, the capital of Oregon, temporarily suspending use of its drinking water treatment facilities that draw water from the 766 mi^2 North Santiam River basin during the February 1996 flood (Bates et al. 1998). X-ray diffraction analysis of suspended sediment in turbid water revealed smectite clay of the montmorillonite group of clay minerals, which forms exceedingly small particles with surface electrical properties that permit them to remain in suspension for many weeks, thus producing very persistent turbidity. Using clay mineral analysis of possible sediment sources within the watershed, it is possible to “fingerprint” earthflows as a major source of turbidity-producing smectite (Bates et al. 1998). Thus, natural geomorphic features (earthflow landforms) and processes (earthflow movement and flooding that causes erosion of earthflows) play a strong role in temporal variability turbidity. The degree to which land use practices affect earthflow movement rate and peak flows in streams eroding the toes of these ancient landslides is debated. These findings of relations between rock/soil types, processes of sediment delivery, and downstream water quality are common in other areas of the Cascade Range in Oregon (Youngberg et al. 1975, Taskey et al. 1978), and the general approach to fingerprinting causes of water quality degradation can be applied more broadly.

The North Santiam River case provides several examples of competing interests within a single watershed. The case is complicated by a large flood-control reservoir in the middle of the watershed which, while reducing flood levels in downstream areas, releases turbid water over a very protracted period, 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 (GAO 1998). While Salem’s water treatment system was shut down due to high turbidity levels, other, superior treatment facilities, such as those of Eugene, Oregon, were treating water with higher turbidity (GAO 1998). The municipal watershed supplying Portland, Oregon, has generally stable soils, but a wet winter triggered a single landslide in an unmanaged area that interrupted water supplies by degrading turbidity. Logging and roads in the watershed have long been quite controversial, but it has been difficult to demonstrate degradation of water quality associated with these land uses. Other basins with more extensive area of unstable rock and soil types, such as the North Santiam River, are likely to have lower water quality, even if land use activities were absent.

Natural processes that severely disturb vegetation, such as fire and extensive wind-toppling of forests, can affect drinking water quality. With regard to wind storms, major concerns in the eastern United States range in scale from localized storms (Hack and Goodlett 1960) to regional hurricanes (Foster et al. 1997). The potential of wildfire to degrade drinking water supplies is a prevalent problem in western mountain landscapes where fire can be an important factor affecting both pulses and long-term patterns of sediment production (Swanson 1981) and nitrogen concentrations in runoff (Beschta 1990). Fire is also prevalent in grassland systems, but its effects on sediment production can be where fire does not kill the vegetation and ground surface roughness may be little altered by fire (Gray et al. 1998, p. 162). Effects of these vegetation disturbances on downstream water quality depend on the severity of disturbance to vegetation and soil, timing of precipitation in relation to vegetation disturbance, and the propensity of the landscape and ecosystem to produce water-quality degrading compounds. However, we know of no studies of directly addressing drinking water quality in response to these processes.

Natural variability of ecosystems is integral to biological functioning and their ability to be resistant and resilient to natural perturbations and management practices. The natural streamflow regime, for example, is tightly coupled in many respects with aquatic biology (Poff et al. 1997). It is also reasonable to assume that temporal and spatial variability in water quality characteristics is integral to the organization and function of aquatic systems. The chemical constituents of water, for example, may set the timing and path of homing by migratory species and periods of turbid water may provide hiding cover. The thermal regime of river systems strongly influences the pace of development of organisms, especially invertebrates, and the rates of many ecological processes, such as decomposition. Thus, modification of the distributions of these water quality characteristics may alter ecosystems.

Reliability of Findings - Existing information is generally good in terms of understanding natural controls on water quality, because of a long history of water use, detection of problems, and studies to build basis for problem solution (e.g., Anderson et al. 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 are generally not reported in terms of instantaneous sample concentrations that are most useful in addressing drinking water quality questions. These and other long-term and short-term studies generally corroborate results of earlier work.

Despite our growing knowledge of natural patterns of water quality, new contexts of drinking water issues are stretching the reliability of existing information. For example, long-term studies of water quality from small watersheds involve land use treatments that are unlike those being used today. These new practices involve lower intensity of site treatments (e.g., partial vs. clearcutting, lower intensities of slash fires, 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 suppression had been effective, addressed in Section 5. Secondary Links. In these cases, water quality objectives will compete with ecological objectives in the contributing terrestrial ecosystems. Furthermore, water quality standards are changing for a host of reasons, not only for drinking water use, but also to meet refined ecological objectives and to supply high technology companies, which may not want water subjected to standard chemical treatments for drinking water.

These factors of an evolving social and biophysical context of drinking water issues suggest the need to reveal the limits of knowledge and possibly take a risk assessment perspective in tackling emerging drinking water issues.

Ability to Extrapolate Findings - Present knowledge pertains to the specific geophysical and biological conditions of the study sites. 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 et al. 1992). These data compilation efforts occur, for example, in the emerging Forest Service's National Water Initiative and is a common factor in many bioregional assessments (Johnson et al. 1999).

Secondary Linkages - Interactions of natural features and processes with past and current management practices is an important issue in evaluating potential future water quality conditions. In the future, it will be challenging to reconcile management actions to meet water supply objectives as well as the objective of maintaining and restoring biological diversity and natural ecosystem dynamics, including maintenance of natural disturbance and flow regimes. 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 reduced longer-term risks of greater risk of high-severity fire under higher fuel loads in the future. Issues such as this pertain in the full spectrum of land uses from intensive plantation forestry to wilderness.

Ability to Address Issues - Land managers and policy makers have available generally good knowledge of the influence of natural processes on water quality, based on a long, strong history of watershed science, including at many experimental forests managed by the USDA Forest Service around the country. Important challenges are emerging in cases such as the North Santiam River watershed where understanding of ecological, geophysical, and human factors must be integrated over large watersheds. Bases for carrying out this integration are being developed in watershed analyses conducted through dam reliscencing 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 - Research needs fall in four categories.

1. Studies are needed of key processes that affect drinking water quality. The target processes may vary among ecological, geological, and climatic settings across the country. In the North Santiam River, for example, refinement of understanding of the relative contributions of different processes of development and delivery of smectite and stratification of turbid water in the reservoir that collects and releases water of poor quality would improve the basis for evaluating and managing future water quality.

2. Watershed-scale assessments are needed of sediment sources operating during and after extreme events. It is important to better quantify the significance of these important events by maintaining long-term studies, good water quality monitoring 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 bring their skills to the job and to foster development of science at this geographic scale and scope of interdisciplinary work.

3. There is need for enhanced basis for extrapolating findings from process studies and assessments of individual watersheds of intensive study to broader scales, such as regions. This sort of "regionalization" analysis is occurring in a variety of management and research sectors on topics relevant to drinking water quality, but not specifically on that topic. Relevant tools (i.e., geographic information systems), analysis approaches, and data bases are available.

4. Good record keeping by managers of water treatment facilities would provide researchers with much improved data bases for evaluating long-term trends in water quality from watersheds used as drinking water sources. Though not a research need itself, this is an important step in ultimately furthering research into causes and cures of water quality problems.

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Atmospheric Deposition

E. S. Verry and J. A. Lynch

In some locations atmospheric deposition may add to other soil and human factors that pose a health risk to domestic water supplies. The constituents of most concern are high hydrogen levels (pH values $\leq$ 4.5 in precipitation), high nitrate levels (> 10 mg/L of nitrate nitrogen (NO_3-N) in soil water or groundwater), and the interaction of these in soil water to yield high concentrations of aluminum (Al) and lead (Pb). Additionally, high hydrogen levels in supply water can dissolve Pb in solder joints where copper pipes are used for plumbing. Treatment of municipal water supplies should be adjusted to a pH range of 6.5 - 8.5 (U. S. Environmental Protection Agency, 1998). However, administrative sites that draw on snowmelt, shallow well, or cistern water are not routinely tested for pH (it is not a water quality standard), and may not routinely adjust water pH.

Areas of Low Precipitation pH and Low Buffering Capacity in Soils

When water supply sources have a pH of $\leq$ 4.5 and the water distribution system contains soldered pipe joints (copper pipe), exposure of Pb to children (0-6 yr.old) can exceed 60 µg/day. Blood levels in young children average 5-10 µg Pb/dL. Exposures exceeding 60 µg/day of Pb, will yield blood concentrations of 10-15 µg Pb/dL that are associated with adverse health effects in young children and in the *in utero* blood of child bearing women. Locations where these conditions may occur are associated with water supplies taken from cisterns, shallow wells, and snowmelt-fed surface waters. Exposures of 85 to 95 µg Pb/day can occur at these locations (NAPAP, 1991).

The number of people using cisterns, shallow wells, ponds, or springs in the eastern United States is estimated at 132 million. However, the sites at risk are those where precipitation pH is $\leq$ 4.5, and where surface soils do not contain enough bicarbonate bases (calcium and magnesium bicarbonates) to neutralize the precipitation acidity. Additionally, in parts of the coal region, some sites can use only cistern water sources because local groundwater is already extremely acid (pH < 4.0) from acid mine drainage. Areas in the United States where precipitation pH averages $\leq$ 4.5 are restricted to the Upper Midwest and Eastern United States and are shown in figure 4.1. Acid surface soils are likely to occur in the areas shown in figure 4.2. These soil areas correspond to areas where lakes and streams are acid, and thus shallow groundwater is assumed to be acid (Church, 1983). Water supply sources taken from surface waters that occur where these two maps overlap should be tested for pH at the faucet, and discontinued until the pH is adjusted between 6.5 and 8.5. Surface water source chemistry can be strongly affected by large rains or snowmelt thus sampling at these times will reflect potential problems best. Be aware that areas outside the map areas can also have low pH precipitation (e.g., sandstone ridge tops).

Aluminum and cadmium, can also cause adverse health effects; however, there is insufficient data to document that atmospheric deposition actually causes concentrations in soil or groundwater to exceed the lowest observed adverse effect level (LOAEL) (NAPAP, 1991). Atmospheric deposition is a primary source of mercury (Hg) and it also can cause adverse health effects. However, its adverse health form is delivered as methyl mercury that is bio-concentrated in fish tissue rather than in the water supply.

High Nitrate Levels in Surface Waters

High nitrate concentrations in drinking water can cause serious and rare occasions of fatal poisoning in infants less than 3 months old. Susceptibility for some occurs at nitrate nitrogen concentrations above 10 mg/L, yet many infants have drunk water containing nitrate nitrogen > 10 mg/L without development of oxygen starvation in the blood (methemoglobinemia) (Walton 1951, Winton et al. 1971). In the United States a drinking water standard not to exceed 10 mg/l of NO_3-N (or 44 mg/L NO_3) is recommended (U. S. Environmental Protection Agency 1972, 1998). Causes of high nitrates in drinking water are usually associated with nitrogen fertilizer use or septic tank contamination of wells. Administrative and domestic sites with shallow wells, and over-loaded septic systems are candidates for high nitrogen levels in well water. (e.g., rural areas where former district ranger residences with an independent well and septic system are converted to day care facilities under special use permits). Analysis of the tap water for nitrate nitrogen to confirm NO_3-N concentrations < 10 mg/L is required at these public sites. This caution is also true on a larger scale where agricultural irrigation (containing nitrogen fertilizer) has pervaded forest areas containing forest administrative sites, and where groundwater and stream water concentrations of nitrate nitrogen exceed 10 mg/L (MPCA 1998).

Inorganic nitrogen in precipitation rarely exceeds 2 mg/L of ammonium nitrogen (NH_4-N) or 3 mg/L of nitrate nitrogen (NO_3-N). Common concentrations are 0.2 and 0.5 mg/L of N, respectively. However, ammonium can convert to nitrate nitrogen in soil water, and concentration of soil water by evapotranspiration (about threefold) can yield soil water concentrations up to 15 mg/L of nitrate nitrogen if it is not consumed by microbial or vegetation uptake. In most cases, this nitrogen is readily taken up by vegetation and soil water nitrogen concentrations remain low. However, forest soils can become nitrogen saturated and yield high nitrate concentrations in soil water and surface water fed streams (Fenn et al. 1998).

Most forests are nitrogen limited and strongly retain nitrogen delivered in rain and snow. Even nitrogen saturated forests (in the Appalachians, the Adirondacks, the Colorado Front Range, or PNW mountain sites with coarse and base-poor soils) do not exceed 1.5 mg/L NO_3-N in soil or stream water, and single-site water supplies in these areas are not a health risk. In contrast, soil water in chaparral watersheds of California, where dry deposition of nitrogen is high, regularly exceeds 3 mg/L of NO_3-N. When these watershed burn severely, nitrate concentrations in soil and stream water can rise to 15 mg/L NO_3-N (Fenn et al. 1998). Administrative sites in these locations with independent well and septic systems may be subject to high nitrate levels in their water supply. Other nitrogen-saturated sites are less susceptible to severe wildfires, and no studies document what happens to nitrogen concentrations in soil water when fire does occur.

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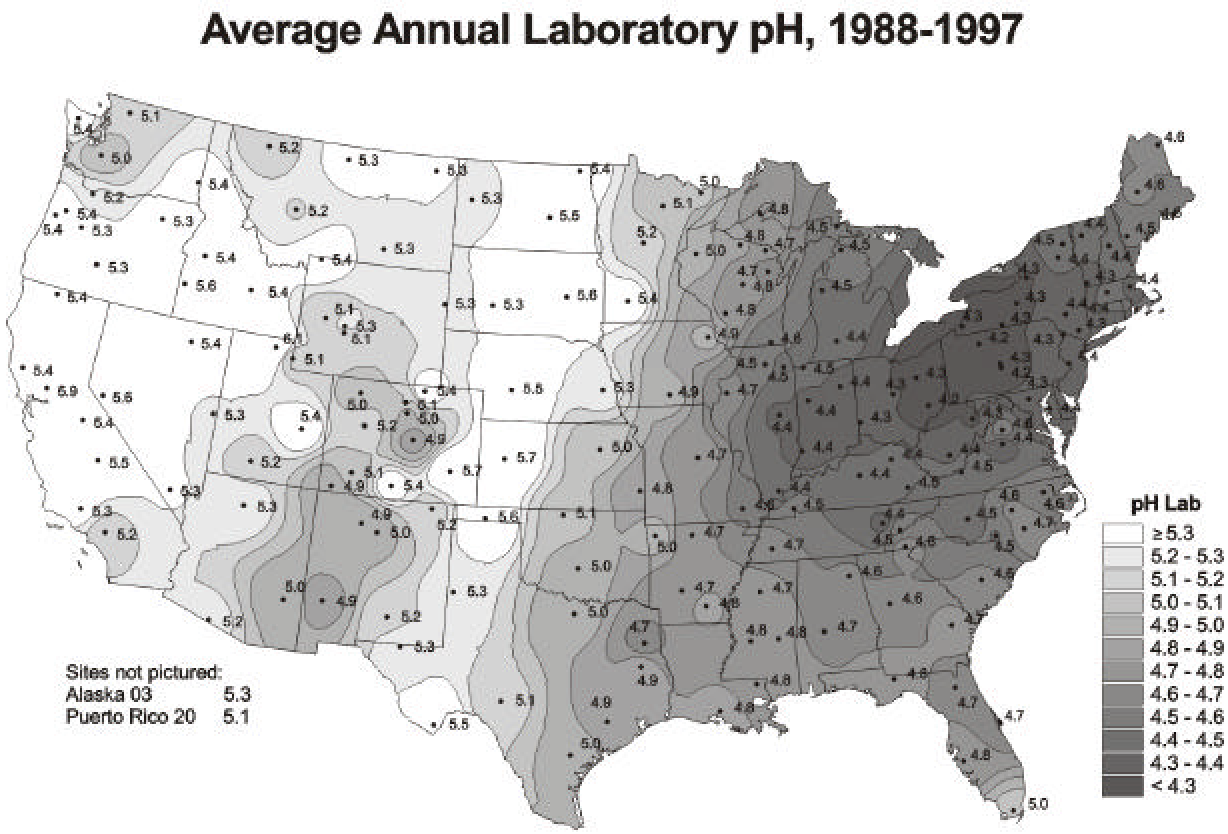

Figure 4.1. Annual average laboratory pH values of precipitation in the United States, 1988 - 1997 (NADP 1999).

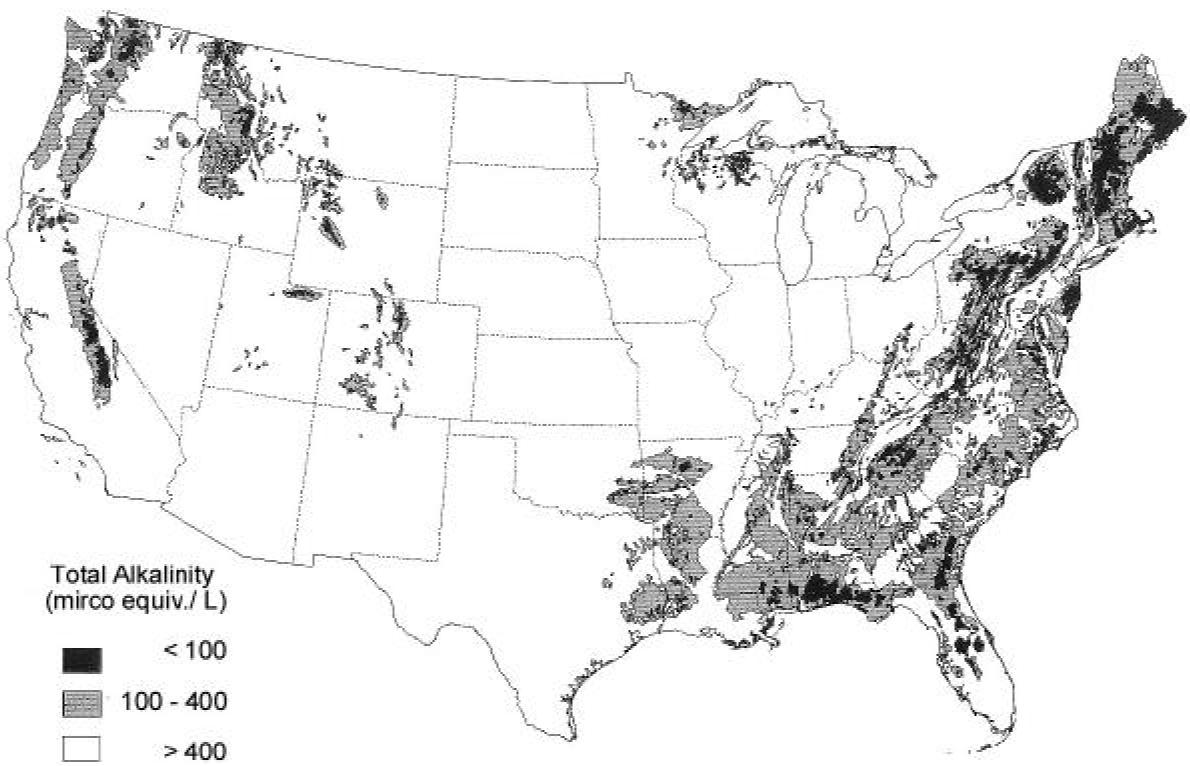

Figure 4.2. Total alkalinity in lake water (and by inference surface soil water) for the United States. Grey areas have lakes with an alkalinity from 100-400 •eq/L and white areas are higher. Black areas contain lakes with an alkalinity less than 100 •eq/L. Where lakes have negative alkalinities within this these black areas, surface water pH values may be less than 4.5. Map prepared by J. M. Omernick, G, E, Griffith, J. T. Irish, and C. B. Johnson (U.S. EPA Environmental Research Laboratory, Corvallis, Oregon).

Forest Succession and Water Quality

Wayne Swank

The effects of forest management activities on water quality are generally of the greatest magnitude in the first several years following disturbance (Chapter 5). However, during long-term succession and recovery of forest ecosystems, changes in physical, chemical, and biological parameters of streams may occur. Changes in stream inorganic chemistry and sediment yield were observed over a 20-year period following clearcut cable logging of a 146 ac. southern Appalachian watershed (Swank et al., In Press). Stream solute concentrations and nutrient fluxes showed small increases after harvest and responses were largest the third year after treatment, except for NO_3^-. There was an initial increase in NO_3^- from < 0.1 mg l^{-1} to 0.8 mg l^{-1} (Fig. 4.3) and an increased net N export of 1.16 lb ac^{-1} the third year after harvest. However, later in succession (15-20 years), NO_3^- 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 woody decomposition. Streamwater concentrations of other solutes showed little change in concentrations (NH_4^+, PO_4^{3-}) after cutting or only small increases (Ca^{2+}, K^+, Mg^{2+}) with subsequent decline in concentrations later in forest succession.

Other long-term research in eastern forested watersheds (Edwards & Helvey 1991; Swank & Vose 1998; Flum & Nodvin 1995) show that as forests mature, less nitrogen is retained in the watershed and stream NO_3^- concentrations increase. These long-term studies support findings on shorter-term successional chronosequence or synoptic 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 NO_3^- and K^+ concentration. Streams draining old-aged forests had higher concentrations of NO_3^-, K^+, and other solutes than did streams draining intermediate-aged forests at the same elevation. Spruce-fir watersheds with no record of logging had streamwater NO_3^- concentrations of about 3 mg l^{-1}, while spruce-fir watersheds logged 30 years previously had NO_3^- concentrations < 0.5 mg l^{-1}.

Another survey of 38 streams draining partially or entirely clearcut watersheds was conducted in New England (Martin et al. 1985) in northern hardwood sites in new Hampshire, Maine, Vermont; central hardwood forests in Connecticut; and coniferous forests in Maine and Vermont. Streams draining partially or entirely clearcut watersheds in the previous two years were selected. There were no apparent changes in stream nutrient concentrations from many of the ecosystems and the largest concentration increases were for NO_3^-, Ca^{2+}, and K^+ in northern hardwoods of New Hampshire. Inorganic N (NO_3-N plus NH_4-N) increased to an average of 2 mg l^{-1} (Martin et al. 1985). However, elevated solute concentrations appear to be short-lived even in streams draining successional northern hardwood forest in New Hampshire (Horbeck et al. 1987). Moreover, early stream chemistry changes following clearcutting were considered insufficient to cause concern for public water supplies or for downstream nutrient loading (Martin et al. 1985).

An extensive synoptic water quality assessment was conducted for numerous streams in the Great Smoky Mountain National Park in the southern Appalachian Mountains (Silsbee and Larson 1982). Concentrations of NO_3^- in streams draining watersheds, which had been logged prior to park establishment, were significantly lower (one-half) the NO_3^- concentrations in unlogged watersheds at similar elevations. The magnitude of stream NO_3^- concentrations associated with long-term forest succession depend on a number of factors such as levels of atmospheric nitrogen deposition, the type & rapidity of forest regrowth, soil microbial activity, and soil physiochemical reactions. However, stream NO_3^- levels rarely exceed 5 mg l^{-1} and are below current drinking water standards, although NO_3^- may contribute to stream acidification, particularly during spring snowmelt events in the northeastern U.S. when peak NO_3^- concentrations occur (Murdock and Stoddard 1992).

Similar to NO_3^-, stream sediment may also exhibit long-term dynamics following 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. The clearcut cable logging study on the southern Appalachian watershed entailed only three contour access roads since logs could be yarded 1000 ft. with the cable system (Swank et al., In Press). Record storms (15 in) in the last two weeks of May 1976, prior to grass getting established, 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. 2a). In the latter two weeks of May, sediment yield was nearly 60 tons from 0.21 ac. of road contributing area (roadbed, cut, and fill). In the ensuing period of road stabilization and minimum use (June-Dec. 1976), soil loss was low but accelerated again briefly during the peak of logging activities (Fig. 4.4a). In the next year, soil loss below the road declined to baseline levels.

Sediment yield at the base of the second-order stream (Fig 4.4b, weir site) draining the watershed showed different temporal patterns than 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 and 1980, there was a cumulative increase of 240T in sediment yield (Fig. 4.4b). During the next 10 years, sediment yield declined with a cumulative increase in export of 240T. The rate of sediment yield over the 5-15 year period after disturbance was about 300 lb ac^{-1} yr^{-1}, or 50% 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 watershed. 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 pre-disturbance 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 exceptional storm events.

The long-term effect of forest harvesting practices alone appears to have little deleterious impact on some water quality parameters (sediment and chemistry) of primary concern in drinking water. However, there is some evidence of management legacy (sediment) and solute dynamics (NO_3^-) associated with succession that may be important to evaluate in a context of cumulative sources from other land uses.

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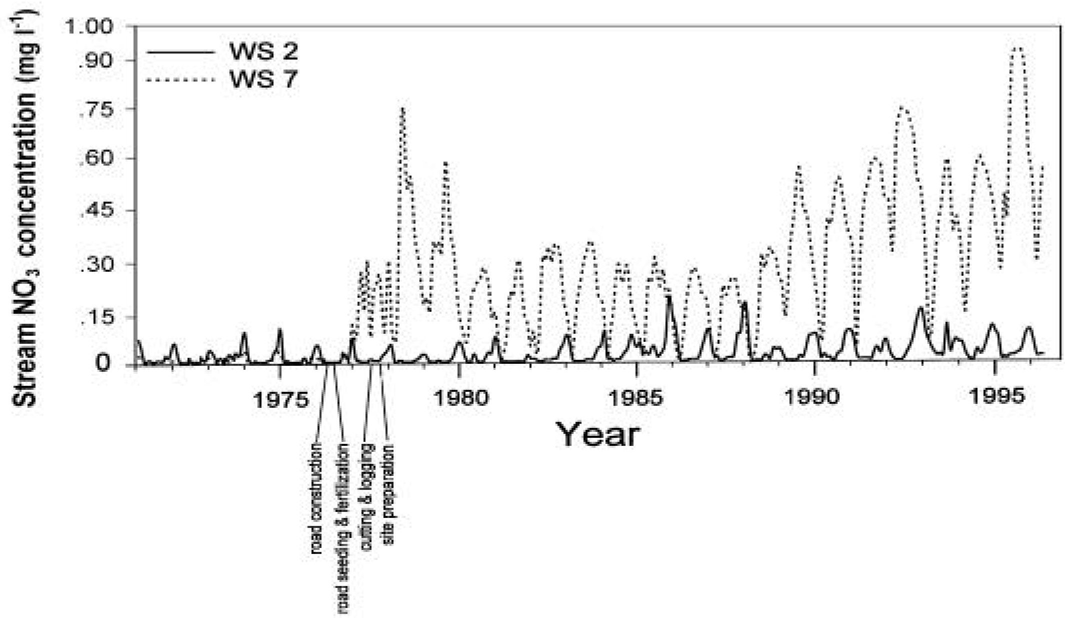
Figure 4.3. Mean monthly concentrations (flow weighted) of NO_3^- in streamwater of a clearcut, cable logged, hardwood covered watershed (WS7) and an adjacent control watershed (WS2) during calibration, treatment activities, and post-harvest period, Coweeta Hydrologic Laboratory, NC.



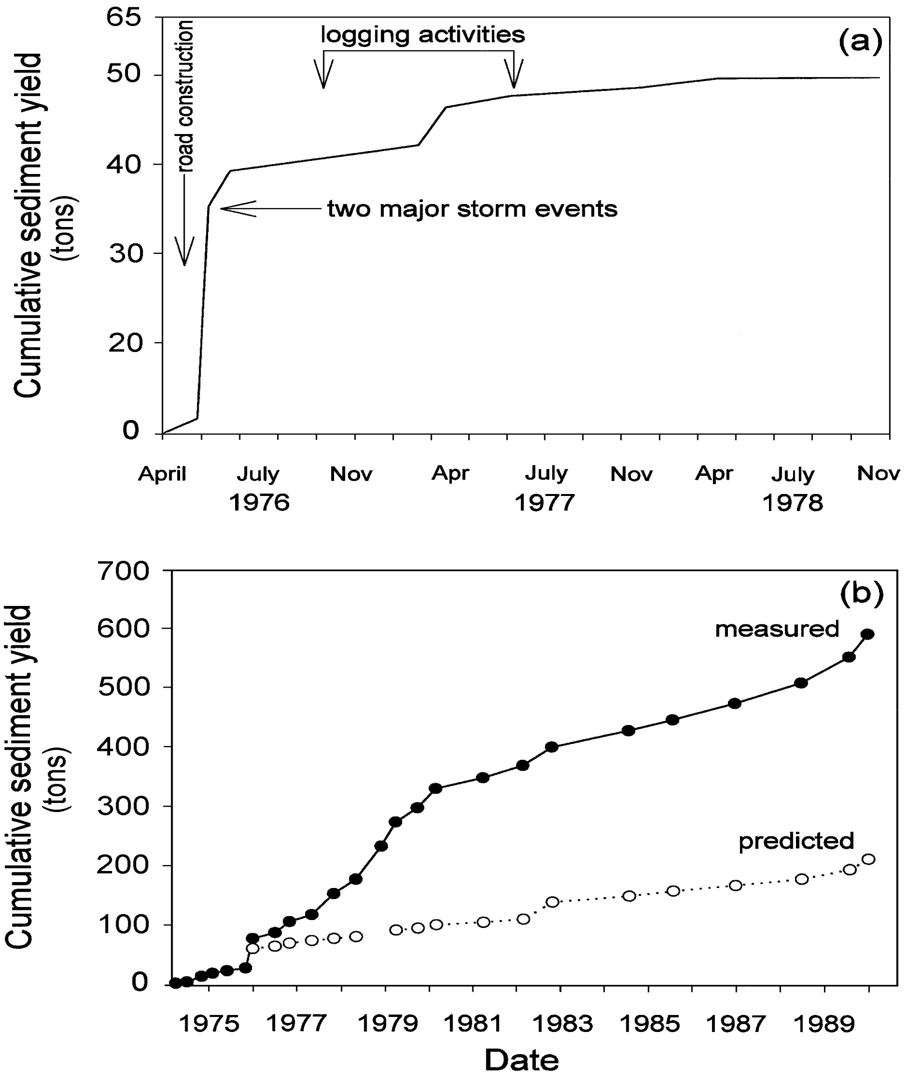


Figure 4.4. Cumulative sediment yield measured on a clearcut, cable logged, hardwood covered watershed: a) in one of the first-order streams below a logging road during the first 32 months after treatment and b) in the ponding basin of the second-order stream at the gaging site during 15 years after treatment. Predicted values are based on pre-treatment calibration of sediment yield with an adjacent control watershed, Coweeta Hydrologic Laboratory, NC.

Chapter 5

Vegetation Management

Forest and grassland vegetation is managed for a variety of purposes, which can affect drinking water quality. This chapter addresses four areas of vegetation management: timber, grazing, fire, and pesticides.

Effects of Timber Management Activities in Water Quality

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Timber management activities have the potential to affect water quantity as well as water quality. Concerns about the potential effects of forest management activities on water quality are not new. Early interest in the effects of forest harvesting on water yields, particularly peakflows or floods resulted in the designation of experimental forest catchments throughout the United States.

In general, the first forest hydrology studies examined the relationship between forest cover and streamflow. These studies were not process oriented, but rather an inventory of the effects of forest cover removal on streamflow timing and yield. The first catchment experiment in the United States began in 1909 at Wagon Wheel Gap in Colorado. This study measured streamflow before and after timber harvesting on a treatment catchment as compared to a control catchment (Bates and Henry, 1928). Syntheses of catchment studies on annual water yield have been completed (Bosch and Hewlett, 1982; and Stednick, 1995).

These paired catchment studies were later used to determine the effects of forest management activities on water quality. A treatment catchment would be purposely impacted to allow detection of water quality changes. These land use treatments could result in water quality changes and help illustrate processes that control water quality . Forest management activities designed to minimize water quality effects often incorporate these controlling processes.

Forest management activities include harvesting trees, removing trees from the site, forest stand regeneration, and stand improvement. Soil disturbance from tree felling is minor. Movement of the log or whole-tree to a landing or collection point may disturb the soil surface. Other soil surface disturbances are related to collection and haul roads. Roads are addressed in separately in Chapter 10. Stand improvement may include selective harvesting of trees, either dominant crown positions or suppressed crowns. 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 5.3) or by herbicides (see Chapter 5.4).

This chapter will review the potential effects of timber management on water quality. Potential water quality effects from timber harvesting or management include sediment, stream temperature, and nutrients.

Erosion/Sedimentation

Introduction - 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 stream sedimentation with potential decreases in site and stream productivity. The following definitions are offered to help the reader: soil erosion is the detachment and movement of soil particles, measured as Mg per ha per yr; sediment is the transport of eroded soil materials by streamflow, measured as a concentration mg per L; and turbidity (which is often a water quality standard) is an optical measurement of the water's ability to defract light, measured as Nephelometric Turbidity Units or NTU (Stednick, 1991).

Soils and geology are the foundation of any forest ecosystem. Soil properties which affect erosion processes are of primary consideration. These properties include vegetative cover, soil texture, soil moisture, and slope among others.

The sediment load of streams (both suspended and bedload) is determined by such characteristics of the drainage basin as geology, vegetation, precipitation, topography, and land use. The sediment enters the stream system by erosional 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 silvicultural or other land use activities upstream.

Issues and Risks: The forest practices with the greatest potential for causing erosion and stream sedimentation are road related and intensive site preparation. Some forest soils on steep slopes may be subject to large mass soil movements (see Chapter *). Other soil erosion processes may occur at smaller scales and rates. Generally, as site disturbance increases, soil erosion may increase.

Most soil erosion studies only measure the amount of soil moved or displaced. Soil erosion measurements include soil pins or other slope survey mechanism, downslope collection devices, in-stream sediment traps or back calculations from sediment concentrations measured in the surface waters. The actual amount of soil erosion reaching the surface water is a small percentage (2 to 10) of the erosion occurring in the watershed. The most common linkage is the sediment delivery ratio, the amount of sediment produced divided by the amount of soil erosion as a function of the watershed area (see Dunne and Leopold, 1979).

Soil erosion and sediment generation may result in graphic effects. Decreased site productivity may adversely affect forest productivity. Further, site revegetation or delay in site revegetation may allow continued soil erosion and sediment production. Sediment accumulation in stream channels may adversely affect water quality and aquatic life. Stream sedimentation may adversely affect stream macroinvertebrates, intergravel dissolved oxygen, and hyporehic flow and migration paths. When waters with increased sediment or turbidity are used for drinking water, treatment costs will increase. The water must be filtered or stored to remove suspended sediment. Often chlorination rates must be increased to disinfect the water since bacteria may be associated with the sediment.

Findings from Studies: Forest practices have the potential to affect water quality in a number of ways (Sopper, 1975). Accelerated sedimentation has been identified as the primary water quality problem related to forest management activities. Activities associated with timber harvest, yarding, and road construction and maintenance result in disturbance of soils and vegetation and may increase soil susceptibility to erosion. These disturbances may alter the hydrologic cycle, generating more overland flow for detachment and transport of sediment.

Undisturbed and forested watersheds usually have erosion rates from near 0 to 0.5 Mg per ha per year (Binkley and Brown, 1992). Erosion rates have been estimated as less than 0.25 Mg per ha per year for three quarters of the eastern and Interior West forests (Patric et al., 1984). Typical timber harvesting and road construction activities may increase erosion rates to 0.1 to 0.5 Mg per ha per year (Table 5.1). More intensive site preparation treatments such as slash windrowing, stump shearing, or roller chopping may increase soil erosion rates up to 10 Mg per ha per year.

Numerous studies have been done on the effects of different forest management practices on erosion rates or sediment production (Table 5.1). In general, increased site disturbance will result in increased soil erosion and subsequent production. The type and magnitude of erosion depends on the amount of soil exposed by management practices, the kind of soil, steepness of the slope, weather conditions, and any treatments following disturbances (Swank et al., 1989).

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

Yarding methods used for the movement of logs from the stump to a landing can be classified as tractor, cable, aerial or animal. Tractor skidding is accomplished with either crawler or wheeltype 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 use of crawler tractors (Davis, 1976; Bell et al., 1974). One method of decreasing the amount of soil disturbed by crawler tractors or wheeled skidders is through careful layout of skid trails (Rothwell, 1971). Planning for skidroad location and number 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 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 et al., 1976; Davis, 1976; Stone, 1973). Helicopters and balloons will likely result in minimum site disturbance, but both are costly and subject to operational constraints.

Logging in the Southeast United States increased erosion to 4 Mg per ha per year from the undisturbed 0.01 Mg per has per year, site preparation attributed to 10 percent of the increases. (Hewlett, 1979). Intense site preparation of 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 10 Mg per ha per year (Pye and Vitousek, 1985).

Any increase in sediment yields from timber management activities is usually short-lived. Surface soil disturbances provide a sediment supply, but once the finer transported materials are moved, that site is less apt to continue eroding. Sediment yields or measured suspended sediment concentrations decrease over time as a negative exponential (Leaf, 1974; Beschta, 1978; Megahan, 1980; NCASI, 1999). This time factor should and can be used in land use planning (Stednick, 1987).

Erosion pavements can form quickly on some soils in the west. But in the south, many surface soils are fine texture to depths of several inches to several feet. The soil surface often becomes sealed, accelerating surface runoff, erosion and sedimentation processes. Fine soil particles continue to be transported by surface runoff until the area is completely revegetated, which may be 2 years for logging, 3-5 years for skid trails and temporary logging roads, and 3-5 years depending on the type of site preparation practice.

Some form of site preparation is often needed to insure the establishment of the tree reproduction following 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 seed bed, and control of competing and non-desirable vegetation. Site preparation includes fire, herbicide, slash windrow, roller-chopping, soil disking or other mechanical techniques. Fertilizer applications may be used to help establish seedlings, but applications usually are applied after tree establishment.

In the Southeastern United States, upland hardwoods may be converted to pine stands. Site preparation includes burning, or chemical treatments to kill the existing vegetation. Soils in the region are often fine textured and deep, thus soil erosion pavements may not develop on the soil surface and increased erosion may last many years. A winter burn and herbicide application increased stormflows, overland flows, peakflows, and sediment production from two small watersheds in Northern Mississippi (Ursic, 1970). Most of the hydrologic effects were evident three years after the fire when monitoring ended.

Sediment transport is a function of stream energy, the areal extent and channel proximity of soil disturbances. Sediments are often finer textured materials with large surface areas. These large surface areas are reactive and may sorb (adsorb and absorb) various water quality constituents including phosphorus, introduced chemicals, and petroleum products. Since phosphorus has a low solubility, phosphorus exports may be associated with sediment transport or export.

Sediment routing and storage are particularly important components in the transport process of sediment loads through the stream system. They are critical to the quantification of short- and long-term impacts of land use activities on stream channels and beneficial uses. However, the storage and routing processes are highly variable and exhibit non-steady state behavior. Continuously evolving relationships involving sediment supply and energy availability are used to estimate sediment discharge changes as the result of land use activity.

Catchment studies have identified correlations between annual peak discharge and annual sediment discharge, and between total annual flow and annual sediment discharge (NCASI, 1999a). Hydrologic modification may upset channel stability, increasing turbidity and sediment concentrations.

Standards and guidelines used by the Forest Service include establishment of 30-meter (100-foot) buffer strips along all water courses (USDA Forest Service, 1981). Riparian strips have standards and guidelines for basal area reduction, surface disturbance and slash disposal practices (USDA Forest Service, 1981).

Landslides have been attributed to forest management activities. These landslides have implicated decreased root strength after tree harvesting, or road prism or road drainage failures. Road related problems are discussed in Chapter 11.

Reliability of Findings - Most studies have shown that increased site disturbance has the potential to increase soil erosion and sediment production. Soil erosion from undisturbed forest watersheds is low. Site disturbance from timber harvesting activities will vary by logging and yarding techniques, site preparation methods, operator techniques, soil vegetative cover, soil moisture, slope, soil depth, and soil texture among other environmental factors.

Ability to Extrapolate Findings - Soil erosion processes are well understood and applicable across the United States. This understanding has been used to develop various soil erosion prediction models that may have regional applicability.

Secondary Linkages - Soil erosion may occur with road construction, use, and maintenance, as well as recreational activities such as off-highway vehicles or trail use by foot or animal. Roads and trails are site disturbances that often have long lives, both may concentrate overland flow and exacerbate soil erosion (see Chapter 11). Since phosphorus has a low solubility, sediment transport may represent phosphorus transport.

Ability to Address Issues - Soil disturbance may result in increased soil erosion and subsequent suspended sediment. Increased suspended sediment concentrations are often graphic, whether there is or is not land use activity. Forestry or grazing operations are often blamed. Poorly designed and/or implemented forest practices have resulted in water quality impacts, particularly with regard to sediment or turbidity. Suspended sediment movement is usually energy dependent. That is higher streamflows are required to initiate and maintain the sediment transport. Suspended sediment may be transported from an 'undisturbed' watershed given a sufficient streamflow.

Sediment concentration (and turbidity) measurements are relatively straightforward. However, soil erosion on a watershed is variable in time and space, and the eroded soil must reach the stream channel to become sediment. Once in the stream channel, sediment transport is episodic, depending on the streamflow. Variations in streamflow make for variations in sediment transport rates. Sediments may be stored in the channel and released over a longer period of time. In-channel disturbances may create in-channel sediment sources, separate from the hill slope processes.

Increased sediment inputs to stream channels may be better assessed by monitoring the physical features of the channel (State of Idaho, 1987; MacDonald, et al., 1991). Such features include channel width, depth, pool volumes, substrate size, and large woody debris among others.

Forest practices that are planned and implemented to minimize site disturbance will not adversely affect water quality. Site disturbances when hydrologically isolated or located away from surface waters, have minimal areal extent, are quickly revegetated, and disturbed in the 'dry' season will have the least effect on soil erosion and suspended sediment.

Research Needs - Soil erosion research is often conducted by using soil erosion plots. An area of land, usually some fraction of an acre (0.001 to 0.1) is hydrologically isolated by a border, usually metal sheeting driven into the soil. The downslope plot edge has a trough or some other device to collect runoff and eroded soil. The weight of soil collected from that area over a given time interval is the erosion rate, i.e. Mg per ha per year.

There is no standard or protocol for erosion plot research. Erosion plots are variable in size, edges may affect water flow, trough collection efficiency changes as sediments are collected. The trough itself may arbitrarily control base level and hence erosion on the plot. Erosion plots integrate erosion over time. An unpublished erosion study in Oregon, showed that the bulk of annual soil erosion occurred in the 24-hour period immediately after the fire. The erosion was as dry ravel, and water erosion or overland flow added little. Precipitation events or storm energies are rarely reported, and comparisons to a control plot may or may not be included. A standard research method for soil erosion studies should be decided upon. The importance of dry ravel as an agent of erosion needs further investigation, too.

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 it is personnel and equipment demanding. Do suspended sediment concentrations best measure the effects of site disturbance? Monitoring stream channel form and function, particularly as related to aquatic habitats, may provide a better metric of land use effects, particularly when assessing multiple or cumulative effects. Annual water yield increases from timber harvesting have been documented, while the effects of timber harvesting on peakflows is less clear. Can this hydrologic modification increase sediment transport from in-channel sources and result in channel morphology changes? Conversely, how much increased sediment input can a stream segment receive without a channel morphology change? Research efforts need to address cumulative effects on channel form and function.

Sediment and erosion studies have identified the watershed level and plot level processes, however there are few studies that address the linkages at the watershed level. New sediment water quality criteria are being proposed that may move soil erosion/sediment monitoring away from the question of optical turbidity or suspended sediment concentrations. A recent TMDL (Total Maximum Daily Load) decision uses the mean particle size (D50) as a measure of stream health. The percentage of fine sediment has been related to the survivability of salmonid fry emerging from stream gravels; as fines increase the probability of emergence and survival decrease.

The ecological effects of sediment on fisheries, particularly salmon, have been studied in the Pacific Northwest. The timber-fisheries linkage has not been investigated in other areas.

Stream Temperature

Introduction -In general, water temperature is not important in terms of drinking water unless significant temperature increases occur. Increased temperatures decrease dissolved oxygen concentrations, which may affect aquatic health, particularly cold water fish species. Warmer stream temperatures may increase algae growth rates and other eutrophication processes, i.e. organic matter decomposition, plant respiration, etc.

Water temperature is an important factor regulating aquatic life and biological and chemical processes (Stednick and Gilbert, 1998). The biologic metabolic rate doubles for every 10°C change, but the saturated dissolved oxygen concentration will decrease by 20 percent. Stream temperatures depend on a variety of energy transfer processes, including radiation, evaporation, and water temperature of subsurface flows entering the stream (Sullivan et al., 1992; NCASI, 1994). Seasonal variations in stream temperature generally follow the seasonal pattern of air temperatures lagged by approximately one month.

The impact of timber harvesting on stream temperature was not addressed until the 1960's. Watershed studies investigated the effects of timber harvesting on changes from water yield and timing, and suspended sediment concentrations and/or production. Fisheries studies addressed chronic and acute toxicity of water temperatures on various fish species and life stages in laboratory settings.

Observed decreases, particularly salmon runs in the Pacific Northwest led to investigations on the effects of timber management on stream temperatures. One of the first studies to investigate the effects of timber harvesting on stream temperatures was the Alsea Watershed Study in Western Oregon which helped initiate legislation of state forest practices acts and development of management practices to prevent significant changes in fish-bearing streams (Beschta et al., 1987).

Issues and Risks - The potential effect of timber harvesting or riparian vegetation removal to increase stream temperatures is recognized. Increased temperatures may adversely affect domestic water quality and stream aquatic life. Harvesting without streamside vegetation understandably has resulted in some rather dramatic temperature increases (i.e., Brown and Krygier, 1970).

Findings from Studies - Surprisingly few studies have been published on the effects of silvicultural practices on water temperature, and many of these were conducted in the 1970's (Table 5.2). These studies include research and management studies, that is harvesting with and without streamside vegetation buffers. Several recent synthesis papers suggest that few additional temperature studies have been conducted (Beschta et al., 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. Other mechanisms, including increased air temperature, channel widening, soil water temperature increases, and streamflow modification have been proposed (Ice, in press). Small streams, with lower surface areas, may be more susceptible to heating, but also recover more rapidly (Ice, in press, Andrus and Froehlich, 1991). Maintenance of streamside shade via streamside vegetation buffer zones can be used as a management tool to avoid temperature increases.

The literature compilation of studies investigating the effects of timber harvesting on stream temperatures (Table 5.2) suggests several points. Most of the stream temperature studies were conducted from 1969 to the early 1980's. From this data set, daily maximum stream temperatures increased from 1.5 to 8°C in the eastern forests and 0.6 to 10°C in the western forests. However, the range in temperature increases also reflects a range in streamside vegetation buffers - from no buffer to a 100m buffer.

Removal of forest canopies over streams may increase fish populations or biomass, either from water temperature increases or increased food chain production or availability (Gregory et al., 1987; Bisson et al., 1992a). Changes in minimum stream temperatures (during the winter or dormant season) range from no change to less than 1°C in the east and from zero to less than 2°C in the west (Table 5.2).

Reliability of Findings - Studies of water temperature changes following timber harvesting have shown temperature increases, decreases , or no change. Stream temperatures may increase when streamside vegetation is removed. The effects of streamside buffers on regulating water temperature are recognized, but several studies have reported temperature increases with streamside buffers. The lack of 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 is recognized but often not considered in monitoring programs.

Ability to Extrapolate Findings - Parameters 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 (i.e., Quigley, 1981 and Amaranthus, no date).

A simpler model developed to predict the effect of clearcutting on stream temperatures uses the calculated heat load to the stream surface area (Brown, 1970). This or similar models should have universal applicability. It would be difficult to suggest one streamwater buffer model as suitable for all forest watersheds, however measurement of the angular canopy density can determine the importance of a buffer strip to prevent stream temperature increases following 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 et al., 1987).

Secondary Linkages - Stream temperature may also be affected by roads or fire.

Ability to Address Issues - Generally, forest practices that open the stream channel to direct solar radiation are the practices which increase stream temperatures. Most aquatic organisms have optimal temperature ranges and temperature increases of more than 2 degrees C may alter development rates of aquatic organisms. Retention of streamside vegetation appears to mitigate potential temperature changes, especially the greater temperature changes. Streamside vegetation buffers need to be evaluated for large woody debris recruitment, stability-both channel meandering and vegetation windthrow, and included in land management/timber planning. The streamside vegetation cannot be left to manage itself.

There is renewed interest in the potential effect of land use activities on stream temperatures as related to water quality standards and fish populations or fish habitat. Accurate stream temperature assessments vary from a single instantaneous measurement to continuous measurement, depending on the stream diurnal and seasonal variations. Similarly, water quality standards (or criteria) may be expressed as a single temperature value not be exceeded (often based on the stream classification), or as the seven-day moving average. Stream temperature data evaluation needs to be evaluated over the long term. Statistical methods include harmonic analysis, time series, and trend analysis (Limerinos, 1978; Hostetler, 1991, Stednick, unpublished).

Research Needs - Streamside canopy removal may also decrease water temperatures, since radiation losses may be increased. The effect of temperature decreases on the stream ecosystem is unknown. Similarly, diurnal variations in temperature may increase after harvesting, the biological effect is unknown, and the quantification of the temperature change limited.

Stream temperature monitoring and reporting protocols need to be developed.. Timber harvesting with proper streamside vegetation buffers should result in none to minimal stream temperature changes. More studies need to be conducted and published documenting the efficiency of streamside canopy cover on stream temperatures.

Few water quality related studies have assessed cumulative watershed effects. Temperature measurement studies at different spatial scales need to be conducted. Finally, long term temperature data are needed to assess the potential effects of single and multiple timber harvesting activities on stream temperature and 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.

Nutrients

Introduction - Water quality from forested catchments is typically higher than waters draining from any other major land use (USEPA, 1995). For example, in the United States water quality from agricultural areas has nitrogen and phosphorus concentrations approximately nine times greater than forested areas (Devernik, 1977 as cited by Binkley et al., 1999). Forested catchments are often used as drinking water sources because of their noted clean waters. Forest management activities such as forest cutting and harvesting interrupt the natural cycling of nutrients and there is concern regarding increased nutrient concentrations.

Timber management practices include harvesting. The greater the harvest or utilization level, the greater potential of nutrients, namely nitrogen and phosphorus to be lost from the system. Nutrient concentrations are typically greater in soil solutions than streams. Nutrient pulses or flushes to streamwaters are usually short-lived and are effectively diluted downstream. Nutrient export or site productivity maintenance is important and downstream uses have not been affected by nutrient exports from timber harvesting.

Nitrogen

A considerable part of the total nitrogen of the earth is present as nitrogen gas in the atmosphere. The oxidation and reduction of aqueous nitrogen species are closely tied to biological activity, and both the paths followed and the end products of such reactions depend very strongly on kinds and numbers of biota present. Small amounts of nitrogen are present in rocks, but nitrogen is concentrated to a greater extent in soil and biological material as organic nitrogen. In general, the oxidation of organic nitrogen in air can be expected to produce nitrite and finally nitrate. Some anaerobic organisms reduce nitrate nitrogen and can produce nitrogen gas instead of, or in addition to, ammonium. In groundwater, nitrate usually appears to be the only form of nitrogen of significance, although nitrate-bearing water may occur in reducing environments (Hem, 1985).

Nitrogen in the form of dissolved nitrate is a major nutrient for vegetation, and the element is essential to all life. Certain species of bacteria in soil, especially those living on roots of legumes, and the blue-green algae, and other microbiota occurring in water, can extract nitrogen from air and convert it into nitrate. Some nitrate occurs in rainwater, although the belief that a significant fraction of this is produced by lightning discharges now seems incorrect, especially given increased NO_x concentrations and acid precipitation formation. Nitrate in the soil that is utilized by plants is partly returned to the soil when the plants die, although some nitrate is lost from the cycle in drainage and runoff and appears in river water.

In natural waters, nitrogen exists as nitrate, nitrite, ammonium, and organic nitrogen (in order of decreasing oxidation state). All of the above forms are biochemically interconvertible, forming the nitrogen cycle (with nitrogen gas). Nitrate is found in small amounts in fresh domestic wastewater, but may attain high levels in groundwater. In nature, nitrates are formed by bacteria that change organic nitrogen to nitrites (NO_2^-) and quickly to nitrates (NO_3^-) under aerobic conditions. Nitrate (NO_3^-) nitrogen is the most completely oxidized state of nitrogen found in water.

Phosphorus

The most common rock mineral in which phosphorus is a major component is apatite, a general name of general species that are principally calcium orthophosphate and that also contain fluoride, hydroxide, or chloride ions (Hem, 1985). These minerals are widespread in both igneous rock and in marine sediments. When apatite is attached by water, the phosphorus species released probably recombine rather rapidly to form other minerals or are absorbed by hydrolyzate sediments, especially clay minerals, in soil. Phosphate is made available for solution in water from several kinds of cultural applications of phosphate in the activities of humans, and these pollution sources probably are the most important causes of high concentrations of phosphate in surface water.

Other phosphate sources include phosphate fertilizers. Phosphate fertilizer application, especially when coupled with irrigation, is a potential source of phosphate in drainage waters. Orthophosphate concentrations increase as sediment concentrations increase. The suspended sediment material as colloidal particulates may carry sorbed phosphate. Irrigated agricultural fields may be large non-point-sources of orthophosphate from fertilizers and soil erosion.

Phosphate ions form complexes with many of the other solutes present in natural water. A calculation of solubility controls over phosphate activity is difficult when so many solute species containing phosphorus would be possible and the form of solids that might be produced is uncertain. Phosphate contents of natural water are not generally restricted by precipitation of a sparingly soluble inorganic compound. More likely, phosphorus uptake by aquatic vegetation and perhaps the adsorption of phosphate ions by metal oxides, especially ferric hydroxide, are the major factors that prevent concentrations greater than a few tenths or hundredths of a milligram per liter from being present in solution in most waters (Stednick, 1991).

Issues and Risks - Nitrate-nitrogen (NO_3-N) is the most oxidized form of nitrogen occurring in natural waters (Hem, 1995). Nitrate-nitrogen (NO_3-N) is an intermediate product of the nitrification process, the oxidation of protinaceous or other nitrogen containing organic matter. Nitrite- nitrogen is potentially toxic, however, nitrite-nitrogen is quickly oxidized to nitrate-nitrogen. Nitrite-nitrogen concentrations are typically below detection levels in forested catchments. Ammonia (NH_3) is also potentially toxic. However, ammonia rarely occurs in natural waters, as the largest sources of nitrite- and nitrate-nitrogen are wastewater treatment plants. Ammonium (NH_4^+) is yet another form of nitrogen in natural waters. Ammonium is not considered toxic, and concentrations in natural waters are usually at or below detection levels since ammonium may oxidize to nitrate, be retained on a cation exchange site, or utilized on-site as a plant nutrient.

Nitrate is a required nutrient for plant life, and too much can lead to eutrophication of lakes and streams since it is often a limiting nutrient in plant growth. High levels in drinking water can cause infant methemoglobinemia (blue baby syndrome). This has led to the maximum allowable limit of 45 mg L^{-1} nitrate or 10 mg L^{-1} nitrate nitrogen for public drinking-water supplies. Nitrate (NO_2^-) nitrogen occurs as an intermediate stage in biological decomposition of compounds containing organic nitrogen. Bacteria converts ammonium nitrogen to nitrites under aerobic conditions, certain bacteria can reduce nitrates to nitrites.

Nitrate nitrogen concentrations are usually quite low (0.002 - 1.0 mg/L NO_3-N) is undisturbed forest catchment surface waters (Binkley and Brown, 1993). This is due to the rapid utilization of nitrogen by ecosystem biota and nitrate formation (nitrification) inhibition. Slow rates of organic matter decomposition, acid soil conditions - common in forest environments, and bacterial allelopathy all decrease rates of nitrification.

The current drinking water quality standard is clearly adequate for protecting human health (Wogan et al., 1996). Nitrate concentrations between 100 and 300 mg N per L are allowed for livestock use (CAST, 1974; and National Academy of Sciences, 1974).

Growth of aquatic vegetation, such as algae, may be influenced by the availability of nitrogen and/or phosphorus. Dense, rapidly multiplying algal growths or blooms sometimes occur in water bodies that periodically receive increased concentrations of nitrogen or phosphorus. These dense growths are generally undesirable to water users and may interfere with other forms of aquatic life, especially if the waterbody becomes overloaded with oxidizable debris as a result of the sudden dieback of an algal bloom (Hem, 1985). The large amount of oxygen required in the organic matter decomposition may result in an oxygen deficit.

The enrichment of a waterbody with nutrients is termed eutrophication and is accompanied by a high rate of production of plant material in the water. Troublesome production rates of vegetation presumably can only occur when optimum supplies of all nutrients are present and available.

Most aquatic systems are phosphorus limited, thus any phosphorus input increase may increase aquatic productivity. Increased aquatic plant productivity may result in eutrophication. Eutrophication is excessive or nuisance plant growth in surface waters with associated water quality changes. Notably, these water quality changes usually occur in lotic systems and include odor, excessive aquatic plant growth, reduced visibility, dissolved oxygen depletion, nutrient releases and reuptake, and sedimentation of dead plants that require more oxygen for decomposition, and potential fish kills from low dissolved oxygen.

There are no water quality standards for orthophosphate-phosphorus (PO_4-P) or total phosphorus (P), and phosphates are not toxic to humans (or animals) unless they are present at very high concentrations. There are no criteria for phosphorus, however recommendations include:

- no more than 0.1 mg/L for streams which do not empty into reservoirs,
- no more than 0.05 mg/L for streams discharging into reservoirs,
- no more than 0.025 mg/L for reservoirs.

Forest soils are usually the largest nutrient pool with biomass as the second largest. The stored quantities of each nutrient varies with climate, species, age, stand density and soil type (Morris, 1992). In most undisturbed forested watersheds, nitrogen and phosphorus exports are less than atmospheric inputs. Precipitation measured as depth and constituent concentrations in the precipitation can be used to calculate nutrient inputs as kg/ha/yr (pounds/acre/yr). Similarly, streamflow expressed as a quantity can be used with concentrations of selected constituents to calculate an export as kg/ha/yr. Cations such as calcium, magnesium, sodium, and potassium have a net export due to parent material weathering.

Findings from Studies - Research conducted throughout the United States has found that nutrient losses following silvicultural activities are usually minimal and do not result in water quality degradation.

Nutrients contained in organic matter: 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. Nutrients associated with the eroded soil particles and sediment may be lost from the site (Swank et al., 1989).

Streams may show symptoms of overenrichment but there is usually minimal opportunity for a buildup of these nutrients in the stream system because of the normally brief period of increased nutrient flux to the stream (Currier, 1980). Other nutrients rarely cause water quality problems, thus 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 to affect downstream water uses or reduce forest site productivity (Swank and Johnson, 1994). Catchment studies have produced a large body of information on streamwater nutrient responses, particularly from clearcutting (Table 5.3). Changes in streamwater nutrient concentrations vary substantially between localities, even within a physiographic region. In Central and Southern Appalachian forests, NO_3^-, K^+ and other constituents increased after harvesting, but the changes were small and did not affect downstream uses (Swank et al., 1989). Clearcutting in northern hardwood forests may result in large increases in concentrations of some nutrients (Hornbeck et al., 1987). Process research on catchments has identified some of the reasons for varied ecosystem response to disturbance (Swank and Johnson, 1994).

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

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 streamwaters
than other constituents. A possible exception, is the loss of Ca and K, documented in the northeast United States when precipitation inputs had higher acids from fossil fuel combustion (Federer, 1989).

Reliability of Findings - The literature compilation suggests a consistent and predictable response in water quality from timber harvesting activities. Total phosphorus concentrations are essentially unaffected by timber harvesting activities. Any increase in ammonium- and nitrate-nitrogen is short-lived and increased concentrations do not represent a decrease in site productivity.

Ability to Extrapolate Findings **-** The rather consistent streamwater chemistry response to timber harvesting allows response extrapolation.

Secondary Linkages - Phosphorus may be associated with suspended sediments and transported off-site.

Ability to Address Issues - In general, forest harvesting practices do not significantly increase phosphorus or ammonium nitrogen concentrations in surface waters. Documented increases in phosphorus or ammonium nitrogen are usually short-lived and always below any water quality standards or criteria. Some forest types in the United States have few studies investigating the influence of forest practices on water quality. Perhaps the perception is that there exists good to excellent water quality in an area and forest practices have little effect on surface waters or water related resources (Tecle, 1991; as cited by Binkley and Brown, 1993b). This could be attributed to level topography, soils with vegetative cover that results in little erosion, soils with infiltration capacities greater than precipitation intensities, low runoff or intermittent streams, or small or infrequent forest practice activities.

Timber harvesting practices may affect nitrate-nitrogen concentrations in soil water and streamwaters. Often cited as an example of timber harvesting effects on water quality is the Hubbard Brook study (Likens et al., 1970). In this study, vegetation was cut and left on-site, and sprayed with a general herbicide for three years to kill any plant regeneration. Nutrient concentrations, particularly nitrate-nitrogen showed significant increases. This study helped identify nutrient cycling processes, but does not represent the effects of timber harvesting on water quality.

Nitrate-nitrogen concentrations may increase in streamwater following timber harvesting, but these concentrations are short-lived as vegetation reestablishes and usually does not represent a threat to water quality or site productivity. There are a couple of possible exceptions. Nitrate-nitrogen concentrations tend to increase in precipitation with increases elevation, noted in areas with air quality concerns (Silsbee and Larson, 1982; Riggan et al, 1985). If timber harvesting occurs in these areas, the higher nitrate inputs probably would result in higher nitrate outputs.

In the Pacific Northwest, water quality samples from forest with nitrogen fixing alder may have higher nitrate-nitrogen concentrations than streams without alder (Miller and Newton, 1983: Binkley and Brown, 1993b). Since nitrogen is being added to the site by fixation this probably does not represent a loss of site productivity, but nitrate-nitrogen concentrations may be high enough to potentially affect downstream uses.

Forest harvesting and site preparation practices that minimize site disturbance and include quick site preparation and forest regeneration establishment seem to minimize any potential water quality effects. Streamside vegetation buffers are effective for sediment removal, but their effectiveness in nutrient retention is unclear.

Research Needs - Soil water chemistry usually has higher nutrient concentrations than surface or stream waters. Changes in water chemistry at larger scales, watershed to landscape need to be evaluated, especially in the context of multiple land use activities or timber harvests for cumulative watershed effects.

Fertilization

Introduction - Fertilization to improve forest stand growth is common. The rate at which nitrogen fertilizer is applied varies by site and timber type, but is usually 224 kg of N per hectare as urea (USEPA, 1980). Phosphorus is supplied at rates between 80 and 100 kg P_2O_5 per hectare in the southeast United States. Fertilizers are usually applied in a dry granular form, thus the major receptor is the forest floor, unless fertilizer is applied directly into surface waters. The use of large and coated urea granules has eliminated problems with aerial drift.

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

Fertilizer chemicals may enter surface waters by several different routes. Direct aplication of chemicals to exposed surface water is the most significant. Identification of surface water bodies to the aerial application essentially eliminates this entry mode. Ammonia absorption by surface waters, when fertilizers are volatilized, is minimal (USEPA, 1980).

Issues and Risks - The issues and risks associated with fertilizer are essentially the same as described in Nutrients.

Findings from Studies - The reported effects of forest fertilization on water quality, particularly nutrient concentrations in streamwaters are variable (reviews by Fredriksen, et al., 1975; Stephens, 1975; Bisson et al., 1992; and Binkley and Brown, 1993b). Nutrient retention by forest soils is excellent and nutrient concentrations in surface waters following forest fertilization are usually low (Table 5.4). Ammonium-nitrogen and phosphorus concentrations are usually well below any water quality standard or criteria. Both of these nutrients are very reactive with forest soils and would be retained on-site. Ammonium-nitrogen concentration may increase in surface waters as a result of direct fertilizer application to open waters. Ammonium-nitrogen concentrations are rapidly reduced through aquatic organism uptake and stream sediment sorption.

Nitrate-nitrogen concentrations measured in surface waters usually peak 2-4 days after fertilization (USEPA, 1980). The magnitude of the peak concentration depends on whether streamside buffers are left along the watercourse, width of the streamside buffer, and the density of small feeder and tributary streams in the streams. Peak nitrate-nitrogen concentrations usually decrease rapidly, but may remain above pretreatment levels for 6-8 weeks. Winter precipitation events may also result in peak nitrate-nitrogen concentrations, but these peaks usually decrease over successive storms and concentrations decrease quickly between storm events.

Reliability of Findings - Relatively few studies have been published on the effect of forest fertilization on water quality. Nonetheless, results generally are consistent and suggest that concentrations of ammonium-nitrogen and phosphorus do not increase after fertilization. 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.

Ability to Extrapolate Findings - Streamwater responses to fertilizer application are well understood and may be extrapolated with reason.

Secondary Linkages - Linkages include other forest chemicals, herbicides, pesticides, or fungicides; see later chapter.

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

Research Needs - The effectiveness of vegetative filter strips or streamside vegetation on fertilizer retention is unclear. Avoidance of direct fertilizer application to surface waters is important, but what else can mangers do to keep fertilizers on-site?

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Table 5.1. Effects of various timber harvests or site preparations on soil erosion.

Location	Treatment	Erosion	Measure	Reference
East				
Georgia Piedmont	control harvesting roading	0.01 Mg/ha/yr 4 Mg/ha/yr 3.6 Mg/ha/yr	sediment	Hewlett, 1979
North Carolina	roller chopping and burn shearing stumps, windrow slash above plus herbicide (glyphosate)	4 Mg/ha/yr 4 Mg/ha 10 Mg/ha	erosion traps	Pye and Vitousek, 1985
Southeast US (?)	natural harvested with roads burned chopped chopped and burned windrow slash disk	0 - 0.5 Mg/ha/yr 0.1 - 0.5 Mg/ha/yr 0.05 - 0.7 Mg/ha/yr 0.05 - 0.25 Mg/ha/yr 0.15 - 0.40 Mg/ha/yr 0.2 - 0.24 Mg/ha/yr 2.5 - 10 Mg/ha/yr	erosion	Burger, 1983
Coweeta, SC	poor road design poor road design good road design good road design w/grass or gravel	400 m^3/km of road 5700 mg/L 1.2 Mg/ha per cm of precip 0.05 - 0.65 Mg/ha	volume/road length sediment concentration erosion per unit precip	Swift, 1988
Hubbard Brook, NH	natural (WS4-WS6) harvest and herbicide	0.025 Mg/ha/yr 0.1 Mg/ha/yr	erosion	Lawrence and Driscoll, 1988
Fernow Exp. Forest, WV	natural harvested	0.017 Mg/ha/yr 0.05 Mg/ha/yr	erosion to stream	Aubertin and Patric, 1974
Cherokee County, TX	natural harvest, chop and burn harvest, shear, windrow, and burn	0.018 Mg/ha/hr 0.013 Mg/ha/yr 0.8 Mg/ha/yr	4 year average	Blackburn et al., 1986 Blackburn and Woods, 1990
Natchez, TN	control harvest only	82 mg/L 183 mg/L	stormflow sediment concentration	McClurkin et al., 1985

Gulf Coastal, MS	control	0.7 Mg/ha/2 yr	erosion over 2 years	Beasley, 1979
	harvest, chop and burn	15 Mg/ha/2yr		
	harvest, shear and windrow	15 Mg/ha/2yr		
	harvest, shear and windrow and plowed beds	20 Mg/ha/2yr		
South Carolina	clearcut	0.15 Mg/ha/yr	erosion	Van Lear et al., 1985
	control	0.02 Mg/ha/yr		
Sumter NF, SC	low intensity burn	0.14 Mg/ha/yr	erosion	Robichand and Waldrop, 1994
	high intensity burn	5.7 Mg/ha/yr		
West				
Fraser Exp, Forest, CO	control	0.04 Mg/ha/yr	to stream	Leaf, 1975
	roads and harvest - Fool Creek	0.01 Mg/ha/yr		Leaf, 1975
	roads and harvest - Deadhorse	0.03 Mg/ha/yr		Stottlemeyer, 1987
H J Andrews, OR	control WS-9	0.03 Mg/ha/yr	to stream	Sollins et al., 1980
	clearcut WS-10	0.2 Mg/ha/yr		
Ouachita, AR	control	0.01 Mg/ha/yr	surface erosion	Miller et al., 1988
	harvest, roller-chop and burn	0.2 Mg/ha/yr		
	harvest, selection cut	0.04 Mg/ha/yr		
Silver Creek, ID	clearcut with buffer	13 Mg/ha/yr - first year 4 Mg/ha/yr - second year		Clayton and Kennedy, 1985
Beaver Creek, AZ	control	0.02 - 0.2 Mg/ha/yr	annual range	Ward and Baker, 1984
	control	24 Mg/ha/yr	observed max	
	31% clearcut	2.9 Mg/ha/yr	observed max	
	100% clearcut	60.9 Mg/ha/yr	observed max	
	clearcut	0.3 Mg/ha/yr	plot erosion	Heede and King, 1990

Location	Treatment	Erosion	Measure	Reference
Caspar Creek, CA	selective clearcut	1.4 m^3/ha/yr		Krammes and Burns, 1973
Drew County, AR	clearcut selection cut control	0.264 Mg/ha/yr 0.013 Mg/ha/yr 0.004 Mg/ha/yr	erosion	Beasley and Granillo, 1988
Clark County., AR	clearcut - mechanical prep clearcut-chemical prep control	0.54 Mg/ha/yr 0.25 Mg/ha/yr 0.07 Mg/ha/yr		Beasley et al., 1986

Table 5.2. Effects of timber harvesting with and without streamside buffers on stream temperature.

Location	Treatment	Maximum Temperature			Reference
		Temperature (°C)	Change (0°C)	Measure	
East					
GA	clearcut w/buffer control	25.0 21.1	3.9	average daily	Hewlett and Fortson, 1982
MD	riparian harvest		4.4 - 7.6	summer max.	Corbett and Spencer, 1975
Coweeta, NC	100% clearcut no buffer control	21.7 18.3	3.4	average daily	Swift and Messer, 1971
NJ	riparian herbicide		3.3		Corbett and Heilman, 1975
Fernow Exp. Forest, WV	95% clearcut w/buffer removed control	16.1 14.4	1.7	average weekly	Aubertin and Patric, 1974
	plot harvesting		4.0	summer max	Kochandorfer and Aubertin, 1975
Hub bard Brook, NH	100% clearcut no buffer control	20.0 16.0	4.0	average daily	Likens et al., 1970
Penn State Forest, PA	riparian harvest		3.9	summer max.	Lynch et al., 1975
Leading Ridge, PA	control 44% clearcut w/buffer	19.4 20.6	1.2	average daily	Rishel et al., 1982
	control 85% clearcut no buffer	17.8 25.0	7.2	average daily	
West					
Alsea, OR	control 85% clearcut no buffer	12.2 22.2	10.0 16.0	average daily summer max	Brown and Krygier, 1970
Steamboat, OR	control clearcut w/buffer	14.4 15.0	0.6	daily max.	Brown et al., 1971
Steamboat, OR	control clearcut no buffer	13.3 15.6	2.3	daily max	Brown et al., 1971
British Columbia	clearcut		0.5 - 1.8	average daily	Holtby and Newcombe, 1982
H. J. Andrews, OR	clearcut		4.4 - 6.7	daily max.	Levno and Rothacher, 1969
Coyote Creek, OR	clearcut		8	daily max,	Harr et al., 1979

Table 5.3. Effects of clearcutting with and without buffers on mean annual nitrate-nitrogen ammonium-nitrogen, and total-phosphorus concentrations.

Location	Treatment	Mean Concentration (mg/L)			Reference
		NO_3-N	NH_4-N	Total P	
East					
Marcell Exp. Forest, MN	74% clearcut control	0.16 0.12	0.55 0.41		Verry, 1972
Hubbard Brook, NH, WS-2	100% cut and herbicide	8.67-11.94	0.04-0.05	0.002	Likens et al., 1970
WS-4	33% strip cut	0.19-0.20	0.05-0.09	0.001	
WS-6	control	0.16-0.29			
White Mountain, NH 7 catchments	controls clearcuts	0.02-0.81 1.31-3.84		0.01-0.02 0.02-0.03	Price et al., 1972
Upper Mill Brook	control clearcut	0.23-0.27 0.23-0.96			Stuart and Dunshie,1976
Leading Ridge, PA, LR-2	100% clearcut and herbicide	0.10-8.4			Corbett et al., 1975
LR-1	control	0.02-0.04			
Fernow Exp. Forest, WV,	100% clearcut control	0.18-0.49 0.10-0.32	0.14-0.35 0.13-0.48	0.04-0.07 0.02-0.04	Aubertin and Patric, 1972; 1974
Coweeta, NC, WS-2 WS-28	control 100% clearcut	0.004 0.094	0.002 0.003	0.006 0.004	Douglas and Swank, 1975
West					
H.J. Andrews, OR	control 100% clearcut	0.020-0.200 0.001-0.010		0.016-0.032	Fredriksen et al., 1975
Bull Run, OR	25% clearcut control	0.002-0.093 0.002-0.013	0.001-0.005 0.002-0.005	0.024-0.039	Fredriksen, 1977
Coyote Creek, OR	100% clearcut control	0.001-0.275 0.001-0.005	0.001-0.018 0.001-0.014	0.011-0.032	Harr et al., 1979 Adams and Stack, 1989
Chicken Creek, UT	13% clearcut control	0.025 0.008		0.014-0.040	Johnston, 1984
Alsea, OR	85% clearcut control	0.19-0.44 1.18-1.21		0.062-0.100	Brown et al., 1973
Priest River, ID	control 100% clearcut	0.20 0.18		0.036-0.060	Snyder et al., 1975
High Ridge, OR	control 40% clearcut	<0.015 0.03-0.1			Stottlemeyer, 1987
Fraser Exp., Forest, CO	33% clearcut control	0.06 0.006			Ryan as cited by Binkley and Brown, 1993
Beaver Creek, AZ	control clearcut	0.010 0.220			

Table 5.4. Effects of forest fertilization on maximum streamwater ammonium-nitrogen and nitrate-nitrogen concentrations (mg/L).

Location	Treatment	NH_4-N	NO_3-N	Reference
East				
Fernow Exp. Forest, WV	258 kg-N/ha as urea	0.8	19.8	Aubertin et al., 1973
West				
Coyote Creek, OR	224 kg-N/ha as urea	0.04	0.17	Fredriksen et al., 1975
Olympic Nat. Forest, WA	224 kg-N/ha as urea	0.18	2.85	Stephens, 1975
	224 kg-N/ha as urea	0.07	0.13	
	224 kg-N/ha as urea	0.22	0.18	
	224 kg-N/ha as urea	0.10	0.07	
	224 kg-N/ha as urea	0.02	0.92	
	224 kg-N/ha as urea	0.10	0.09	
	224 kg-N/ha as urea	0.55	0.40	
Olympic Nat. Forest, WA	224 kg-N/ha as urea	0.04	0.121	Moore, 1975
Entiat Exp. Forest, WA	54 kg-N/ha as urea	<0.02	0.210	Klock, 1971
	57 kg-N/ha as urea		0.068	Tiedemann and Klock, 1973
Mitkof Island, AK	210 kg-N/ha as urea	0.003	2.36	Meehan et al., 1975
Suslaw River, OR	224 kg-N/ha as urea	0.49	7.6	Burrough and Froehlich, 1972
Cascade Mountains, OR	224 kg-N/ha as urea	<0.01	<0.25	Malueg et al., 1972
Chelan, WA	78 kg-N/ha as urea	0.011	0.510	Tiedemann, 1973
South Umpqua, OR	224 kg-N/ha as urea	0.048	0.177	Moore, 1971
Ludwig Creek, WA	200 kg-N/ha as urea	0.004	2.7	Bisson et al., 1992

Water Quality Associated with Rangelands

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Several water quality attributes are associated with livestock grazing of rangeland and with allied land uses. Since livestock grazing is a significant use of rangeland, grazing and management practices that reduce water quality impacts will be discussed. Rangeland management is not simply another phrase for livestock management. Rangelands are a kind of land, because of climate, soils, topography, and/or precipitation characteristics, and are unsuitable to intensive agriculture without irrigation or other intensive managerial inputs. Many rangelands have trees and are grazed.

This paper emphasizes western rangelands. This is appropriate since the traditional U.S. Forest Service permits for livestock grazing tends to be a western scenario. It does not mean to imply, however, that there is no livestock use of other U.S. Forest Service lands. The publications noted herein are meant to reflect physical and biological principles associated with water quality and livestock use. Therefore while specific manifestations of livestock/water quality interactions may vary, the underlying principles should be consistent.

Erosion and Sedimentation

Introduction - Erosion and its consequence, sedimentation, are generally considered to be the number one problem associated with wildland watershed management.

Issues and Risks - With improper grazing, overuse of vegetation will occur. As vegetation is weakened and/or lost, surface erosion rates are accelerated. With increased erosion, soil fertility is lost on-site and high sediment yields are transported off-site. When significant sediment enters stream channels, channels develop point bars deflecting flow in opposite banks and destabilize them, channels widen, etc., and creating additional sources of sediment (USDI. BLM, 1998, USDI. BLM 1993). Sediment changes water clarity, the oxygen carrying capacity of the stream, covers spawning grounds, and changes the biological potential of the stream. Nutrients attached to sediment heighten the possibility of eutrophication.

Findings - Considerable research is available on the relationships between livestock grazing and erosion and sedimentation (Figure 5.1). Several textbooks summarize the effects of livestock numbers, livestock types, timing of grazing, animal distribution on vegetation and erosion (Stoddart, Smith and Box 1975; Holecheck, Pieper and Herbel 1989. Grazing intensity affects vegetation, soil organic material, and infiltration rates, with the higher vegetation density and soil organic material promoting higher infiltration rates (Buckhouse and Gaither 1982, Buckhouse and Mattison 1980). Therefore, grazing management should protect or enhance organic production.

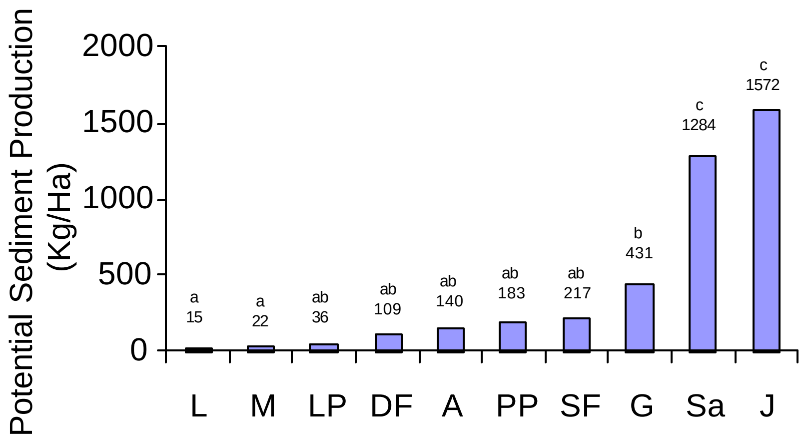

Figure 1. Potential sediment production in 10 Blue Mountain ecosystems. Different lower case letters indicate differences in statistical significance ($P<0.10$) Ecosystems: L - larch, M - meadow, LP - lodgepole pine, DF - Douglas fir, A - alpine, PP - ponderosa pine, SF - spruce-fir, G - grassland, Sa - sagebrush, J - juniper. (Taken from Buckhouse and Gaither 1982)

Figure 5.1 above.

In the riparian zone, the relationship of livestock grazing and streambank erosion has been studied (Buckhouse and Knight 1981, Bohn and Buckhouse 1985, Buckhouse and Bunch 1985). They found it is possible to manage livestock grazing enhance riparian vegetation and protect streambanks. Growing-season-long grazing harms vegetation and results in increased streambank sloughing. Conversely, grazing systems based on plant growth and physiology result in responses statistically similar to non-use. An example of this is an Oregon experience where early season grazing enhanced shrubs, allowing forage harvest without damaging shrubs. (Buckhouse and Elmore 1997).

Jon Skovlin (1984) prepared an excellent compilation of livestock grazing, vegetation, streambank stability, and various water quality issues (Table 5.5). He compiled a large reference list and provides recommendations and prescriptions for grazing management.

Reliability of Findings - This work has been repeated and verified in several locations by several researchers.

Ability to Extrapolate Findings - Conceptually accurate. The key to applying findings is to think in terms of plant physiology and plant response to grazing their affect on sediment.

Secondary Linkages - The key to proper grazing is managing vegetation and soil organic material, thus, the soil's ability to infiltrate, store and slowly release precipitation, and reduce erosion and sediment.

Table 5.5. Outline detailing comprehensive review of rangeland grazing on riparian habitats by Skovlin (1984)

- Introduction
- Function of Riparian Zone
- Values Associated with Riparian Zones
  - Livestock Values
  - Wildlife Values
- Land Use Planning
  - Loss of Riparian Habitat
  - Perspectives on Riparian Zone Use
- Approach to Literature Review
  - Intensities, Specialized Systems, and Other Management Practices
    - Grazing Intensities
    - Distribution
    - Cattle Behavior
    - High Use of Riparian Zones
    - Specialized Grazing Systems
      - Need for Grazing Systems in Riparian Zones
    - Grazing Systems for Public Land
    - Summary of Intensities and Specialized Systems
  - Grazing Effect
      - Vegetation Response
        - Herbaceous Plants
        - Shrubs and Trees
        - Riparian Vegetation
      - Watersheds
        - Infiltration, Runoff and Erosion
        - Water Quality
        - Streambank Stability
      - Terrestrial Wildlife
        - Birds and Grazing Relationships
        - Non-Game Birds
        - Waterfowl
        - Small Mammals
        - Burrowing Rodents
        - Other Organisms
        - Large Mammals
        - Livestock
      - Fisheries
        - Thermal Pollution
        - Habitat Characteristics
- Summary and Conclusions
  - Interaction of Grazing Impacts
  - Interpretation of Literature Reviewed
- Recommendations
  - Prescriptions for Riparian Zone Management
  - Targeting Riparian Zones for Acceptable Restoration
  - Research Needs
    - A Uniform Fisheries Inventory
    - Autecology and Plant Communities Studies
    - Plant and Animal Response Studies
    - Background Erosion Levels
    - Identification of limiting factors
    - Classification and Inventory
    - Non-Game Bird and Small Mammal Research
    - Livestock Production Studies
    - Artificial Propagation Investigations
    - Strategies to Graze Restored Habitats
    - Land Use Planning
    - Regional Socio-Economic Studies
- Appendices
  - A. Glossary
  - B. Classification
  - C. Riparian Resource Allocation

Ability to Address Issues - The concepts are increasingly well known. Success is keyed to site-specific understanding of the ecosystem(s), the vegetation present or desired, and the management objectives, and then prescribing herbivory appropriate to the site. The USFS and the Bureau of Land Management have cooperated to make 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 with remarkable success nationwide (USDI. BLM 1998).

Research Needs - Refinement of herbivory-to-vegetation-to-sediment relationships for use on a site specific/ecosystem basis is needed.

Bacteria and Protozoa

Introduction - Water-borne, pathogenic bacteria and protozoa have long been recognized and have long been an area of public health concern. Many diseases are species specific and can be transmitted among members of the same species. A significant numbers of diseases can be transmitted to hosts of different species. Herein lies the concern about water-borne diseases: Contamination of the water supply by humans and by a whole host of animals poses the potential for widespread hazards to human health.

Issues and Risks - Fecal Bacteria - Fecal coliform bacteria, predominately *Escherichia coli* or *E. coli*, is a ubiquitous bacterium found in the gut of all warm-blooded creatures. 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.

In recent years *E. coli*, which was long considered to be benign, has developed several pathogenic strains which have gained national attention due to their virulent nature.

Drinking water standards remain at > 1 fecal coliform bacterium/100 ml of water. The standard for primary contact water is > 100 fecal coliform bacteria/100 ml of water (appropriate for swimming and accidental ingestion). The secondary contact water's standard is 200 fecal bacteria/100ml of water (water that is less likely to come into intimate contact and ingestion, appropriate to uses such as fishing).

Findings from Studies (Bacteria) - Walter and Bottman (1967) reported on the closed watershed supplying the city of Bozeman, Montana and 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 wild creatures. The increased animal use resulted in the higher fecal bacteria numbers.

Coltharp and Darling (1973) studied three neighboring watersheds with different animals grazing and browsing: wildlife, sheep and wildlife, and cattle and wildlife. Their results 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. 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. None of the three watersheds met drinking water standards (>1 bacterium /100ml of water) and none violated primary contact standards (>100 bacteria/100ml).

Buckhouse and Bohn (1981 and 1983) sampled waters in northeastern Oregon and found similar results. On the watersheds with a resident deer and elk population, where grazing by livestock was excluded for several months during the winter period, the bacteria numbers were low but exceeded drinking water standards. At the end of the summer, after livestock had grazed the area for several months, the bacteria numbers were slightly elevated, but still fell below the primary water quality standards.

Research in central Oregon studied the fate of the coliform bacteria (Moore, Buckhouse and Miner 1988; Larsen et al. 1988; Biskie et al. 1988; Sherer et al. 1988; Sherer et al. 1992; Larsen et al. 1994). They discovered that statistically only five days per year experienced enough precipitation to produce overland flows in this Great Basin rangeland type. Consequently, they reasoned, the probability of washing fecal material into streams was relatively low. In evaluating livestock behavior, a cow will defecate approximately eleven times a day. Less than one of these defecations landed in the stream directly or within one meter of the stream where it might be washed into the stream. They went on to conduct experiments to determine the fate of bacteria defecated directly into the stream. They were able to determine that 90% of the bacteria precipitated to the stream bottom and attached to sediments. By collecting sediment samples over the next several weeks, they determined that 90% of the lodged bacteria had died within 40 days.

They studied the relationship between livestock use of a flowing stream and a strategically placed watering trough in both the summer and winter periods (Miner, Buckhouse and Moore 1992; Clawson et al. 1994). In the wintertime when there was snow on the ground and livestock were being fed hay, a groundwater fed trough was able to attract 95% of the livestock as opposed to only 5% use at the stream. It is speculated that the warmer groundwater (approximately 50°F) held much greater appeal to the cattle than did the 32°F creek. In the summertime, riparian zones tend to provide lush vegetation, shade, and water. A trough simply provided water. As a consequence the livestock showed little preference for the trough until late in the summer when most of the available vegetation near the riparian zone had been consumed. At that point, approximately 25% of the livestock drank from the trough with the remaining 75% still preferring the stream.

Protozoa - Giardia and *Cryptosporidium* have drawn considerable attention recently. Both are debilitating and can be carried by a wide variety of warm-blooded animals. Both have been known for decades, however routine testing following gastrointestinal complaints from citizens have only recently 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 creatures. Calves consistently shed greater numbers of oocysts than do older animals (Atwill 1996). Apparently, by four months of age, calves develop a resistance and the oocysts shed are dramatically reduced. An alternative to livestock exclusion from areas where *Cryptosporidium* may be a concern is to insure that livestock grazing the watershed are older than 4 months.

Reliability of Findings - The *Cryptosporidium* work is a relatively recent endeavor (Tate 1999, personal communication). Tate and his colleagues at the University of California, Davis, continue to investigate these relationships, the biology and the management of *Cryptosporidium*.

Ability to Extrapolate Findings - Conceptually these experiments were solid. Site specific relationships are appropriate to specific biological responses. However, regional climatic and temperature differences may influence the range of variability inherent in these relationships.

Secondary Linkages - No one should think that just because they are in a wildland setting or in a wilderness area, the water is free of biological contaminates. Water purification for recreational uses should be encouraged and promoted as a routine precaution to water consumption.

Research Needs - *Cryptosporidium* research is still in its infancy. Further studies dealing with the origin and fate of the organisms are appropriate.

Impact of Juniper Encroachment and Other Weeds

Introduction - As noted earlier, the test of a functioning watershed is the soil's ability to infiltrate, store and slowly release precipitation. For our purposes, weeds are defined as plants being present beyond the extent of their historical range. Therefore, even a native plant like juniper is a weed when it is present in a habitat where it does not belong evolutionarily. The basic concern is that alien or other aggressive plants are able to colonize sites and change the entire balance of the site. From a watershed and water quality viewpoint, invasive species do not hold the soil as well as the native species, and will likely increase erosion potential and change site productivity.

Issues and Risks - Juniper and other aggressively colonizing plants are a watershed and a water quality problem because of increased erosion and sediment problems. Invading weeds are frequently linked to human activity and/or land uses. A common scenario is that weed seeds are brought into heretofore, isolated locations where no biological control exists for them. Another common scenario is the absence of fire from fire dependent ecosystems, allowing the development of thriving, fire-intolerant plant communities. Since these colonizing species are frequently aggressive and hardy, they are able to outcompete and displace the original vegetation. In water limited systems this frequently results in vegetation composition and/or plant distribution changes.

Findings from Studies - Juniper is a classic example. Historic, abusive grazing pressures around the turn of the twentieth century over consumed fine fuels. In addition, fire prevention policies were initiated and are still being implemented. As a result, fire return frequencies were altered. Juniper is fire sensitive and had been restricted to rocky, shallow soil areas, like rimrocks. With the cessation of wildfire, juniper was able to expand its range downslope. Junipers are aggressive competitors for water and nutrients. They have been able to out-compete and displace much of the understory, herbaceous vegetation.

This resulted in a pattern of uniformly spaced trees fully occupying the site, with bare ground in between. These bare sites have low infiltration rates and are extremely vulnerable to erosion. These inter-spaces lose their A and B soil horizons due to erosion. The erosion rates an order of magnitude higher than what they originally experienced: a 25-year rain event on a sagebrush-steppe community will yield about 100kg/ha in erosion, while an encroaching juniper site will yield 1,000 kg/ha (Buckhouse and Gaither 1982, Buckhouse and Mattison 1980). These sites can be so degraded they cross ecological thresholds and lose their original site potential.

Where juniper has invaded, the sites probably will not carry fire because the understory is gone. Even if they could be burned, there may be nothing to replace the juniper. In these cases, research has shown that cutting the trees and spreading the branches will encourage needles to shed and nutrient cycling to slowly begin. Downed branches also create "safe sites" for understory seed germination and seedling establishment. While expensive, this is the most successful restoration technique (Eddleman et al. 1994). Seeding may be necessary on sites that have lost essentially all of its potential understory seed sources.

Other weed species create similar problems. In most cases the invading weeds have poorer soil holding abilities than do the native species.

Associated with runoff and sediment are phosphorus, nitrogen, bacteria, and oocysts. Water quality is impaired. Overland flow is often warmer than ground water discharges to streams, resulting in increased stream temperature (Stringham et al 1998). Finally, sediment directly impacts a variety of water quality values, including fish habitat.

Reliability of Findings - Strong relationships between weed invasion, native vegetation displacement, erosion, and water quality are well documented.

Ability to Extrapolate Findings - These relationships have wide applicability, but actual water quality responses will vary by situation.

Secondary Linkages - Water quality is an obvious concern as one contemplates weeds and how one might contend with them. Several other compelling aspects are present as well. Wildlife habitat is seriously affected by weed encroachment. Economic values are negatively impacted as alien species crowd out species which may be valuable as a forage base.

Ability to Address Issues - Specific information dealing with the restoration ecology, integrated pest management approaches to controlling weeds, and subsequently establishment of alternative species is continually evolving. The concepts of site specificity and the generalized ecological relationships are solid and workable concepts.

Research Needs - Research needs are related to the details of weed ecology and control. Understanding the plant growth and development processes of each species is critical to be successful in controlling invaders and in establishing ecologically acceptable species.

Chemical and Nutrient Impacts

Introduction - In association with sediment transport are a number of chemical compounds. Most of these compounds are plant growth and development compounds, as opposed to heavy metals and/or toxins. Rangelands are infrequently treated with herbicide and/or fertilizer compounds. The primary chemical/nutrient problems are phosphates (PO_4) and/or nitrates (NO_3) associated with eroding soils and/or direct input of fecal material.

Issues and Risks - Phosphates and nitrates, recycled in the uplands, are essential to plant growth. Potential problems exist with erosion and/or defecation of feces, where PO4 and NO3 find their way to the stream. Excessive levels of these nutrients in streams can produce algae blooms and eutrophication. Algae die-off follows a bloom, usually in late summer, warm water periods. Warm water is less able to adsorb and hold oxygen than is cold water. Oxygen deficiencies create problems for aquatic species during warm weather. If algae die-off corresponds with the warm water period, oxygen is extracted from the water by the decomposition process and the water may become lethally deficient in dissolved oxygen resulting in massive fish kills (Odum 1971).

Findings from Studies - Since much of the NO_3 and PO_4 reaches the stream in association with sediments, see above sections for discussions of overland flow, erosion, and sediment transfer.

The concern about direct deposit of fecal material into the stream in terms of its impact on NO3 and PO4 loading is obvious. However, wetlands can mitigate their concentrations. In an extremely interesting investigation on the Wood River system in Oregon, where nutrient loading studies were conducted on streams originating from the Crater Lake National Park, traversing National Forest, thence private grazing lands, and into a lake listed as hypoeutrophic was recently conducted. Concern had been that nutrient loading would be further increased from water flowing through the livestock grazed section due to contamination of bovine feces. The data did not bear this out. In fact the NO3 and PO4 levels decreased (Hathaway and Todd 1997, personal communication). It was speculated that the system represented a natural "constructed wetland". In wetlands, NO3 and PO4 are reduced via the reduction chemistry processes associated with anaerobic conditions. Further wetland plants take up nutrients from the aqueous system, are eaten by animals and transported away from the water. Thus, these nutrients are not redistributed back into the water when the plant senesces and dies. Perhaps the livestock grazing the “wetland”, consumed the vegetation and its nutrients, and later redistributed the nutrients further from the stream in their feces. In the central Oregon studies it was noted that fewer than 10% of those feces were deposited within a meter of the stream (Moore, Buckhouse and Miner 1988; Larsen et al. 1988; Biskie et al. 1988; Sherer et al. 1988; Sherer et al.

1992; Larsen et al. 1994). It may be that significant amounts of the uptaken nutrients were redistributed away from the stream, resulting in lower nutrient loads in the stream.

Reliability of Findings - The relationships between nutrient loading and sediment is clear. The relationship between the livestock grazing wetlands and the possibility of nutrient loading abatement is not clear.

Ability to Extrapolate Findings - The relationship between erosion, sedimentation and nutrient loading is universal and is one of the bases for erosion control worldwide.

Secondary Linkages - The possibility of livestock being used as a biological tool control weeds, to change plant communities for the better, to promote species which encourage infiltration and reduce overland flow, and even to reduce nutrient loading is exciting. There can be environmentally sound ways to graze; ways which not only provided economic values to our nation, but which can also provide ecological benefits (Bedell and Borman 1997)!

Ability to Address Issues - Most of the concepts discussed in this section are based on solid research and management experience. These concepts are based on the positive relationships between vegetation and organics to enhance infiltration, reducing overland flows and subsequent erosion. The concept of livestock as a constructed wetland, vegetation harvesting tool is intuitively logical, but not exhaustively tested.

Research Needs - Further research into the relationships between livestock harvesting streamside vegetation and nutrient loading of streams is needed.

Overall Conclusions:

In rangelands water quality can be protected by encouraging upland and riparian vegetation which increases the soil's capacity to infiltrate. By understanding the relationships between plant physiology and animal herbivory, one can tailor grazing practices to enhance infiltration. This is usually a function of the timing as well as intensity of grazing. With increased infiltration, sediment, bacterial numbers, nutrients, and temperature relations all respond favorably environmentally. Properly managed, one can have both economic and environmental benefits derived from grazed rangelands.

Fire Management

Johanna D. Landsberg and Arthur R. Tiedemann

When the rains began to fall, few realized what would result from a problem that started with wildfire strategies and ended with Denver=s water supply,@ reported Illg and Illg (1997) on the events subsequent to the Buffalo Creek fire in Colorado. Municipal water supplies were shut off, one of Denver=s water treatment plants was closed, months were spent cleaning a water-supply reservoir, and Coors Brewing Company resorted to bringing in water by truck (Illg and Illg 1997).

Fire, both wild and prescribed, has the potential to alter physical, chemical, and biological properties of forest surface waters that originate from burned areas. Nonpoint source pollution (NPSP) from forests is usually from drainage through the mineral soil or surface flow across the forest floor [Society of American Foresters Task Force on Reauthorization of the Clean Water Act (SAF Task Force) 1995]. Nonpoint source pollution from forests can impair the suitability of water, without treatment, 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 use of fire. Fire management activities (i.e. retardant application, fireline construction) and post-fire rehabilitation (i.e. seeding, fertilization) also have potential water quality implications.

Our purpose is to review our present understanding of the effects of fire, fire management activities, and fire rehabilitation measures on drinking water quality and to examine these effects from the perspective of established drinking water quality standards. The established drinking water standards (Appendix &&) apply after water has been treated [US Environmental Protection Agency (EPA) 1998]. Nonetheless, it is advantageous to prevent the need to treat drinking water supplies where possible. There are two levels of standards: primary and secondary. The primary drinking water standards are legally enforceable standards that apply to public water systems. If water in a public water system does not meet the primary standards, it must be treated. Primary standards protect drinking water quality by limiting the levels of specific contaminants that can adversely affect public health. Secondary standards are non-enforceable guidelines that regulate contaminants that may cause cosmetic or aesthetic effects (EPA 1998).

To accomplish our objectives, we rely on results of research on water quality responses to wildfire, wildfire and fertilization, prescribed fire, forest harvest followed by slash burning, and fire suppression activities.

Sediment and Turbidity

Issues and Risks - Sediment is the major NPSP problem in forests (SAF Task Force 1995). Beschta (1990) reported that sediment and turbidity are the most significant water quality responses associated with fire. Further, these attributes are elusive and difficult to characterize, and both have serious consequences for desirability of water for drinking and other uses (Beschta 1990). The EPA (1992) recognizes causes other than sediment for reduced quality in forest waters, but the effects of those other causes usually are much less common and less severe than those of sediment.

The primary standard for drinking water sediment or turbidity is written in terms of turbidity (EPA 1998), ATurbidity has no health effects but can interfere with disinfection and provides a medium for microbial growth. It may indicate the presence of microbes@ (EPA 1998). Nonetheless, public water systems that use surface water, or ground water under the direct influence of surface water, must meet the standards set in the Surface Water Treatment Rule (EPA 1998). Users of such water are required to disinfect the water and to filter the water. At no time can turbidity (cloudiness of water) go above 5 nephelometric turbidity units (NTU); systems that filter must ensure that turbidity go no higher than 1 NTU (0.5 NTU for conventional or direct filtration) in at least 95% of the daily samples for any two consecutive months.@

Findings - Only two of the studies we found used turbidity as a measure of the sediment content of water (table 5.6). All others used direct measurements of sediment in ppm (table 5.7). The relationship between sediment in ppm and turbidity in NTUs was tested for three catchments in northcentral Washington, and a strong relationship found (Helvey et al. 1985). With this strong relationship and the equations developed, sediment measurements can be converted to turbidity measurements (Fig. 5.2 This allows an approximate determination of NTUs from sediment data. 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.

The two studies that used turbidity measurements are both by Wright et al. (1976, 1982), who measured the effects of broadcast burning on turbidity of streamflow from oak-juniper watersheds in central Texas. They expressed turbidity as Jackson turbidity units (JTU), a measure that approximates NTU (Rand et al. 1975). Watersheds fell into 3 slope steepness categories: 3 to 4 percent, 8 to 20 percent, and 37 to 61 percent. Some turbidity measurements in control watersheds exceeded the drinking water quality standard (5 NTU) before burning, regardless of slope steepness (table 1). Turbidity changes after burning were most pronounced in the steepest watersheds with levels reaching 230 JTU.

Measures of suspended sediment (weight per unit volume of streamflow) (Table 5.7) indicate that fire has a profound effect on this characteristic. Beschta (1980) indicates that a relationship between suspended sediment and turbidity can be established but that the relationship differs significantly among watersheds. He suggests that the relationship must be established on a watershed-by-watershed basis. For Alaskan waters, Lloyd (1987) translated ranges in suspended sediment levels in parts per million (ppm) to a turbidity level. Concentrations of 0 to 25 ppm suspended sediment equates to a turbidity reading of 7; 26 to 80 ppm suspended sediment to a turbidity reading of 19. This relationship is not much help for interpreting the water quality implications of values in table 5.7, since even control or pre-treatment levels in about half of the studies would exceed the drinking water standard. It does point out the need to either standardize the measurements that are made or establish relations between suspended sediment and turbidity for each watershed or stream system in question.

Sediment production, in lb ac^{-1} $yr^{-1,}$ has been measured in many studies because of the concern for soil loss after fire or fire and precipitation or snowmelt. Sediment yield varies widely as a consequence of fire, or forest harvest and fire (Table 5.8). This variability reflects numerous interacting factors: geology, soil, slope, vegetation, fire characteristics, treatment combinations, weather patterns, and climate that influence sediment production.

In the research we evaluated, sediment yield from pretreatment or control areas ranged from as low as 0 lb ac^{-1} yr^{-1} to as high as 4,940 lb ac^{-1} yr^{-1} (Table 5.8). Postburn sediment yield ranged from as low as 12 lb ac^{-1} yr^{-1} to as high as 201,390 lb ac^{-1} yr^{-1}. The lower values generally were associated with flat land and lower severity fires, where fire severity was described. The higher values resulted from fires on steep slopes, and on fires on areas of decomposing granite which eroded readily. These sediment yields are sufficient to generate concern from a water turbidity standpoint, although water turbidity was not directly measured, as well as from a soil impoverishment standpoint.

Seeding of areas that have burned is used to promote rapid establishment of plants and rapid stabilization of the soil resource by the development of plant roots, in hope of reducing erosion and sediment yield. The practice of seeding, and in some cases, seeding plus fertilization after fire are can reduce sediment production from watersheds (Rich 1962, Pase and Ingebo 1965, Tiedemann and Klock 1977). Amaranthus (1989), nonetheless, found grass and fertilizer treatment did not significantly reduce surface erosion compared to the adjacent untreated areas. Pase and Ingebo (1965) seeded one burned chaparral watershed in Arizona to lovegrasses and killed shrub sprouts with 2,4,5-T herbicide. The other watershed was also seeded, but shrubs were allowed to develop. Six years after the wildfire, on the watershed not seeded to grasses, shrubs provided more than 40 percent cover. Nonetheless, sediment yields were lower on the watershed seeded to grasses and treated with herbicide. On the Entiat Experimental Forest in central Washington, seeding improved vegetative cover by 20 to 30 percent compared to the watershed that was not seeded (Tiedemann and Klock 1977). Concurrent with the increase in vegetative cover, sediment delivery rates declined (Helvey 1980). Nonetheless, the stream channels of three of the watersheds were severely scoured by debris flows, and there is no way to determine the role of seeding and fertilization in accelerating the watershed stabilization process. One important observation was that in some areas, the aggressive, perennial seeded grasses provided the only vegetative cover (Tiedemann and Klock 1977). Native species appear to have been effectively excluded. In the short-term, this may have been important for watershed stability, but we don't know the long term successional or watershed stability consequences of excluding native vegetation.

Fireline construction, particularly that which is created by bulldozers, is an important potential threat to sediment and turbidity in streams. First, some firelines are constructed in urgent circumstances, without adequate time to consider stream protection, and thus 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 removed and cast aside. Hence, they are likely to be slow to revegetate with perennial vegetation. Information on revegetating and stabilizing firelines is very limited. Application of seed and fertilizer is an effective way to provide protection to firelines (Klock et al. 1975, Tiedemann and Driver 1983). Klock et al. (1975) demonstrated that several species of introduced and native grasses provided up to 85 percent foliar cover within two years after seeding on firelines. Starter fertilizer with N and S substantially improved plant foliar cover and was considered to be essential for success of such seedings.

With all of the few values we found for turbidity (Table 5.6), including pretreatment values, exceeding the allowable turbidity standard, it is apparent that the effect of fire on turbidity, *per se*, needs further investigation. In face of the large sediment yields after fire and fire related treatments (Table 5.8), it is difficult to imagine that some of these sediment yields would not have produced turbidity values that exceeded the permitted turbidity level. The permitted level applies to water after treatment. The turbidity and sediment yields measured here are before water treatment, and act as an indicator of the need for treatment.

Temperature

Issues and Risks - There are, at present, no standards established for water temperature for suitability of water for drinking. Increases in stream water temperature do have important implications and potential effects on aquatic habitat and on stream eutrophication processes. Eutrophication has the potential to exert secondary effects on drinking water color, taste, and smell.

Findings - When riparian vegetation is removed by fire or other means, the stream surface is exposed to direct solar radiation, and stream temperatures increase (Brown 1970, Gibbons and Salo 1973, Levno and Rothacher 1969, Swift and Messer 1971). These examples will illustrate the potential increases in stream temperature: After clearcutting and slash burning, stream temperatures increased by 13.0, 14.0, and 12,1 °F in June, July, and August with temperatures reaching a maximum of 75 °F in July (Levno and Rothacher 1969). Helvey (1972) found that during the first year after wildfire in eastern Washington, stream temperature increased 10 °F. In southern Oregon, Amaranthus et al. (1989) determined that temperatures increased 6, 11, and 18 °F, from a low temperature of 55 °F to a high temperature of 73 °F after a wildfire. These changes may be important from the standpoint of increasing the rate of eutrophication processes, if phosphate is present in abundance, or of limiting the waters as aquatic habitat.

Chemical Water Quality

Introduction. - Several chemical constituents are likely to come from forest and rangeland sources in the process of burning and are regulated under water quality standards. Primary standards cover N as nitrate-N (NO_3-N) and nitrite-N (NO_2-N), as well as many other substances which are not commonly associated with wildland fire. Secondary standards apply to pH, sulfate (SO_4), total dissolved solids (TDS), chloride (Cl), iron (Fe), turbidity (discussed previously), and several other constituents. Secondary standards are also set for color and odor; phosphorus as phosphate (PO_4) can affect water quality because of its ability to affect color and odor by accelerating the eutrophication process.

To understand the influence of fire on water quality, it is important to elaborate on 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 when critical temperatures are reached, or as particulates carried away in smoke, or released as oxides to the ash layer (Tiedemann 1981, Raison et al. 1984, DeBano 1991, McNabb and Cromack 1990, DeBano et al. 1998) (fig. 2). Of the constituents covered by water quality standards, N, S, and potassium (K)--as a component of TDS--are all susceptible to volatilization loss by burning (Klemmedson 1976; DeBano and Conrad 1978, Raison et al. 1984, Tiedemann 1987, McNabb and Cromack 1990, DeBano et al. 1998). Nitrogen is lost when temperatures reach ~ 400 °F (DeBano 1991, DeBano et al. 1998). At temperatures as low as 700 °F, loss of S can be substantial (Tiedemann 1987). As temperatures approach 1475 °F, virtually all N and S are volatilized. At 1430 °F, P and K are volatilized. Relatively insoluble oxides of metallic cations in the ash such as calcium (Ca), K, magnesium (Mg), and Fe react with water and carbon dioxide of the atmosphere and become more soluble (Tiedemann 1981, DeBano et al. 1998) (Fig. 5.3). This conversion increases potential for leaching loss of nutrients from the ash into and through the soil (Wells et al. 1979, Tiedemann 1981, McNabb and Cromack 1990, DeBano et al. 1998). Nutrients in the ash are also susceptible to loss via surface erosion (Wells et al. 1979, Tiedemann et al. 1979, Tiedemann 1981, Beschta 1990, DeBano et al. 1998).

The potential for increased NO_3-N in streamflow occurs mainly because of accelerated mineralization and nitrification after burning (Vitousek and Melillo 1979; Covington and Sackett 1986, 1992; DeBano et al. 1998), and reduced plant demand (Vitousek and Melillo 1979). The increase results from direct conversion of organic N to available forms (Kovacic et al. 1986), mineralization (Ojima et al. 1988, Covington and Sackett 1992), or mobilization by microbial biomass through the fertilizing effect of ash nutrients and improved microclimate (Koelling and Kucera 1965, Hulbert 1969, Ojima et al. 1988). This effect is short-lived, usually lasting only a year or so (Kovacic et al. 1986, Monleon et al. 1997).

Transport of nutrients to streamflow occurs both during and after a wildland fire. Spencer and Hauer (1991) reported that the source of N in streamwater during a fire appears to be diffusion of smoke gasses directly into the streamwater, and that the source of P in streamwater appears to be from the leaching of ash deposited directly into the stream. After a fire, nutrients move from the soil into streamwater when precipitation is adequate for percolation below the root zone, and when capacity of vegetation for uptake or soil nutrient storage capacity, or both, are insufficient to retain nutrients carried into the soil from ash after fire (Stark 1977, Tiedemann et al. 1979, Beschta 1990, DeBano et al. 1998).

Issues and Risk - The issue is whether forest or rangeland fires will create stream or lake water which is outside the EPA (1988) standards and will therefore require treating to bring it within those standards. Standards for the several chemical entities and properties involved in chemical water quality, as well as the risk of being outside the standard, will be given as each entity or property is discussed.

Findings -

Stream water pH. - The secondary drinking water quality standard for pH is 6.5 to 8.5 (EPA 1998). During the immediate post-fire period, pH of streams may be affected by direct ash deposition. In the first year after fire, increased pH of the soil (Wells et al. 1979) may also contribute to increased stream water pH. In all the studies we evaluated (Table 5.9), only one reported pH values outside the EPA (1988) standards. After the Entiat fires in eastern Washington, Tiedemann (1973) detected transient pH values up to 9.5 during the first 8 months after the fires, and a transient pH value of 9.2 two days after fertilization.

Nitrogen

The primary drinking water standard for NO_3-N is 10 ppm; the similar standard for NO_2-N is 1 ppm. Nitrogen in surface waters from wildland watersheds occurs principally in four forms: dissolved organic N, NO_3-N, ammonium-N (NH_4-N), and urea-N. The combined concentration of these seldom exceeds 1 ppm, and dissolved organic N is usually the most abundant form. No standards exist for dissolved organic N, NH_4-N, or urea-N in drinking water, and they will not be discussed.

We would expect that the magnitude of stream water quality responses to prescribed fire would be less than those observed for wildfires and for some broadcast slash burns. It is unlikely that prescribed fire would consume as much forest floor, understory, or kill as much overstory as the wildfires reported here (McNabb and Cromack 1990, DeBano et al. 1998). Stream chemistry responses to prescribed fire in an undisturbed ponderosa pine/Gambel oak watershed in Arizona (Gottfried and DeBano 1990) provide support for this speculation. Surface fuels were burned on 43 percent of the watershed and 5 percent of the trees were killed. The fire resulted in only slight (but significant) increases in NO_3-N which did not approach the primary standard (Table 5.10). Measures taken to protect streams and riparian areas with unburned buffers could also minimize effects of fire on stream chemistry.

The most striking response of NO_3-N concentration in streamflow to fire, and the only case where the primary water quality standard was exceeded (Table 5.10), was observed in southern California (Riggan et al. 1994). Severe burning resulted in a maximum NO_3-N level of 15.3 ppm in streamflow, compared to 2.5 ppm in streamflow from an unburned control watershed. Maximum concentration for a moderately burned watershed was 9.5 ppm. Results of Riggan et al. (1994) probably represent an unusual response because the watersheds they studied were subject to chronic atmospheric deposition of pollutants. In the other studies we examined, regardless of the treatment or treatment combination (i.e. wildfire and fertilization; clearcutting and slash burning), levels of NO_3-N were below maximum allowable concentrations (Table 5.9). Beschta (1990) reached the same conclusion in his assessment of streamflow NO_3-N responses to fire and associated treatments. It was evident from the results of Tiedemann (1973) and Tiedemann et al. (1978) that fertilization after fire resulted in higher concentrations of NO_3-N than fire alone (Table 5.9). Nonetheless, It is probably safe to conclude that neither fire nor N-fertilization after fire at levels less than 54 lb ac^{-1} of elemental N will have adverse drinking water quality consequences for NO_3-N.

Nitrite-N had been measured in only two studies that we found. In California at the Lexington Reservoir in Santa Clara County, Taylor et al. (1993) found NO_2-N levels of 0.03 ppm after a fire occurred in the watershed feeding the reservoir, while control levels were 0.01 ppm. Tiedemann (1973) reported that NO_2-N concentrations were below the levels of detection.

Nitrogen-containing fire retardants have the potential to affect the quality of drinking water, but research on the application of retardants to streams has focused on effects on fish and the aquatic habitat (Norris et al 1978, Norris and Webb 1989). In an *in vitro* research project to determine the toxicity of some retardant formulations to stream organisms, McDonald et al. (1996) tested two non-foam retardants containing sulfate, phosphate, and ammonium compounds (Fire-trol GST-R[19] and Phos-Chek D75-F), a retardant containing ammonium and phosphate compounds (Fire-Trol LCG-R), and two

[19]References to trade names, commercial products, or manufacturers do not imply or constitute government endorsement or recommendation for use.

foam suppressant compounds which contained neither sulfate, phosphate, nor ammonium compounds (Phos-Chek WD-881 and Silv-Ex). They found concentrations of nitrate-N rose from 0.08 to 3.93 ppm after adding the non-foam retardants. In addition, they found nitrite-N (for which the primary standard is 1 ppm) reached concentrations as high as 33.2 ppm. The solutions they tested were much less concentrated than that which is used in fire-fighting retardant application, but exact concentrations of the retardant test solutions is not given. One of the retardants, Fire-Trol GTS-R, when mixed for fire-fighting, is used at the rated of 1.66 pounds per gallon of water Ba concentration of 200,000 ppm.

Phosphorus

There is no established standard for drinking water quality for PO_4-P. Nonetheless, we think it is important to discuss PO_4-P responses to fire, forest harvest and fire, fire plus fertilization, and direct addition of PO_4-P as a component of fire retardants because of the potential indirect effects on drinking water quality through the eutrophication process. Algal blooms that develop as a consequence of accelerated eutrophication could influence color, odor, and turbidity. Total dissolved PO_4-P occurs at very low levels in streams from wildland watersheds. Prior to fire, PO_4-PO_4-P concentrations ranged from 0.007 ppm to 0.17 ppm (Wright 1976, Tiedemann et al. 1978, Longstreth and Patten 1975, Hoffman and Ferriera (1976), Tiedemann et al. 1988). After wildfire, prescribed fire, or clearcut followed by broadcast burning, PO_4-P concentrations stayed the same or increased only as high as 0.2 ppm (Longstreth and Patten 1975). These increases are probably unlikely to be significant for eutrophication of the aquatic habitat. If the change in PO_4-P concentration as the result of an inadvertent application of retardant directly into a stream has been evaluated, we were unable to find it.

Sulfur

The secondary drinking water quality standard established for SO_4-S is a maximum of 250 ppm (EPA 1998). The sulfate ion is also a relatively mobile anion in soil - water systems (Johnson and Cole 1977). Although not as well studied as N, the mineralization processes for S are similar. In streamwater from wildland watersheds, observed levels of SO_4-S are well below this standard (Table 5.11). Control or pretreatment values range from as low as 1.17 ppm to as high as 66 ppm, while posttreatment values range from 1.7 ppm to a high of 76 ppm, all far below the EPA standard.

Chloride

The secondary drinking water quality standard for chloride (Cl) is 250 ppm (EPA 1998). Chloride response to fire and clearcutting plus fire has been documented in several studies (Table 5.12). Chloride concentrations in control or pretreatment samples ranged from 0.49 ppm to 6.4 ppm, and the chloride concentration in posttreatment samples ranged from 0.40 ppm to 7.1. Interestingly, both the pretreatment and the posttreatment high values were found in Lewis Lake in Yellowstone National Park.

Total Dissolved Solids

The EPA (1988) secondary standard for TDS is 500 ppm. Only two studies reported total dissolved solids; many other studies measured some of the constituents of TDS but not TDS *per se.* Hoffman and Ferreira (1976) detected a TDS concentration of about 11 ppm in the control area and 13 ppm in the burned area, which had been a mixed conifer and shrub area in Kings Canyon National Park, California. Lathrop et al. (1994) found Yellowstone Lake and Lewis Lake, in Yellowstone National Park, Wyoming, had pretreatment TDS concentrations of 65.8 and 70 ppm. The TDS concentration after the fires was 64.8 and 76 ppm. These concentrations do not approach the secondary standard limit.

Trace Elements

Fredriksen's (1971) results raise a question about how well we understand the responses of micro-nutrients or trace elements to fire and fire after clearcutting. In his stream chemistry profile of clearcutting and broadcast burning, he documented a maximum concentration of manganese (Mn) of 0.44 ppm. The EPA (1998) secondary drinking water quality standard is 0.05 ppm.

Several other trace elements, including heavy metals, have established drinking water standards. Information on the effect of these elements after a forest or rangeland fire on drinking water quality is lacking.

Reliability, Confidence, and Limitations - The results of research on the effects of fire on drinking water quality are strong and consistent, especially from the Pacific Northwest, the Rocky Mountain area, and the Southwest. Data from the Southeast are somewhat more limited in spite of the region=s extensive prescribed fire program. In spite of the large number of wildfires in Alaska, water quality research after fire is limited.

The results evaluated for this summary indicate that fire and fire management practices produce findings that are similar for the areas of the United States that are reported. The results are most often in terms of small increases in the substance or entity of interest. The effects of fire and fire-related treatments on water pH and nitrate-N, chloride, and sulfate concentrations appear to be adequately researched. Nonetheless, data are limited for turbidity measured in standard nephelometric turbidity units and for total dissolved solids, to the extent that these results cannot be taken as final.

Ability to Address Issues - The principal issue is: Does fire affect water quality so that the water does not meet the EPA (1988) drinking water standards for a public water supply? The standards apply to water after treatment. Nonetheless, we evaluate the impact of fire and fire-related practices on water quality before treatment.

Existing literature on the effects of fire, fire plus fertilization, and forest harvest followed by fire is substantial. This research, however, was conducted to determine the effects of fire on other ecosystem variables, not directly on water quality. Therefore, some of the elements and compounds for which the EPA (1998) has set standards for drinking water have been little measured or measured not at all. Occasionally components were measured but because the results between pretreatment and posttreatment values were not significantly different, the actual values were not reported. If the values had been reported, they could have been compared to the standards for safe drinking water.

Several gaps in the knowledge base become apparent: fire effects on turbidity measured in standard nephelometric units; effects on nitrite-N production; effects on trace elements; and effects of retardants on water quality.

Additional measurements of fire effects on turbidity measured in standard nephelometric units are needed to adequately evaluate this potentially major effect. Fire and fire-related management practices, and fire followed by sudden, intense precipitation events, produce tremendous sediment yields (Table 5.8), and this sediment load cannot help but impact water turbidity.

Nitrite-N measurements are included in very few reports. One reason for the absence of documentation is that nitrate-N and nitrite-N are often determined as a single entity and the concentration of both is reported as one number. Also, nitrite-N concentrations can be below the limits of detection.

Noticeably absent from the literature on the effects of fire on water quality is an assessment of trace elements, including heavy metals. These elements would not be expected to appear in stream or lake water unless a source for the trace elements were located within or near the watershed. Old mining operations may need to be considered as potential sources if a watershed burns. Fredriksen's (1971) findings that after a fire the concentration of manganese was above the allowable standard, remind us that we cannot assume a trace element is not present without actually running the analysis.

Ability to Extrapolate Results - There is sufficient documentation from the major fire regions of the US, excepting Alaska, that the need for extrapolation is greatly reduced. Extrapolation from the results of similar fires, i.e. wildfires extrapolated to other wildfires, and prescribed fires extrapolated to other prescribed fires, within a geographical region, is fairly safe. Nevertheless, results for which there are few data available must be used cautiously. Several researchers (Tiedemann, personal communication; Beschta 1980) directly caution against extrapolating results of turbidity and sediment research beyond the areas in which the research was conducted. This is especially true for equations which convert from sediment in ppm to turbidity in NTUs - equations which are developed for water from a specific watershed with unique soil characteristics and plant communities. These equations would need to be validated for each new watershed before they can be applied there.

With the results for pH, nitrate-N, sulfate, and chloride so consistently below the EPA standards, a land manager can safely believe that the results will be similar to those found in the literature considered here, if the treatments are equivalent. Nitrate-N concentrations in areas of heavy pollution may not be within the EPA standards and need further evaluation.

Research Needs - Standardized procedures for sampling, measuring, and reporting turbidity are needed. Sediment concentration has been measured in some studies and turbidity in others. Turbidity is related to the amount of suspended sediment (Beschta 1990) and regression relationships can be established between the two characteristics (Beschta 1980, Lloyd 1987, Waters 1995), but this must be done for each individual watershed or stream system.

We also don't have an adequate understanding of the relations of developing vegetative cover, erosion, sediment, and turbidity after fire. How effective is seeded vegetation in the long-term and does native vegetation eventually replace dense stands of seeded species? What are the implications of seeding versus not seeding for long-term watershed stability and water quality?

Areas with chronic atmospheric deposition, such as those studied by Riggan et al. (1994), may need further study of the relations of fire and N release into streams.

An exception to the adequacy of our understanding of stream chemistry and fire relations is that we have little information on abundance of trace elements (micronutrients) such as lead, copper, fluoride (Fl), Mn, Fe, zinc, and mercury, among others, in water from wildland watersheds. Nor do we understand effects of fire in combination with other treatments on micronutrients. This may be particularly important for some of the trace elements with established primary drinking water quality standards such as Pb, Cu, Hg, and Fl.

The inadvertent application of fire retardants directly into a stream could have water quality consequences for NO_3-N, sulfate, and possibly trace elements. Information about these potential effects of retardants on water quality is limited, is important, and is needed.

Literature Cited –

Table 5.6. Water turbidity after fire only or fire in combination with other treatments can reach levels higher than the EPA standard which states, A At no time can turbidity (cloudiness of water) go above 5 nephelometric turbidity units (NTU); systems that filter must ensure that turbidity go no higher than 1 NTU (0.05 NTU for conventional or direct filtration) in at least 95% of the daily samples in one month@ (EPA 1988).

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Author(s) year
Prescribed fire,	Juniper	Central Texas			Wright et al. 1976
pile and burn		3 to 4% slope	12[a]	12	
		8 to 20% slope	20	53	
		37 to 61% slope	12	132	
Pile and burn.	Juniper	Central Texas	12[a]	162	Wright et al. 1982
Pile, burn, and seed				72	

[a] Jackson turbidity units.

Table 5.7. Suspended sediment concentration in streamflow after fire only or fire in combination with other treatments; there is no EPA (1988) standard for suspended sediment concentration.

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Author(s) year
			- - - - - - - - - - - - ppm	- - - - - - - - - - - -	
[a] Cited in Wildfire	Taiga	Interior Alaska	3.7 to 10.6	2.6 to 6.0	Lotspeich et al. 1970
Clearcut, slash broadcast burned	Douglas-fir	Western Oregon	2	56 to 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 et al. 1982
Pile, burn, and seed			1.0	3.7	
Prescribed fire	Loblolly pine plantation	Upper Piedmont, South Carolina	26 ppm	33 ppm	Douglass and Van Lear 1983
	Wildfire	Lodgepole pine,	Glacier National	<3	15 to 32 Hauer and Spencer 1998
	Douglas-fir, ponderosa pine, western larch	Park, Montana			

[a] Maximum value attained.

Table 5.8. Sediment yield after fire only or fire in combination with other treatments.

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Author(s) year
			--------------------lb ac^{-1} yr^{-1}--------------------		
Annual burn	Oak woodland	Mississippi	50	660	Meginnis 1935[a]
Annual burn	Woodland	Guthrie, Okla.	20	220	Daniel et al. 1943[a]
Prescribed fire	Hardwood woodland	North Carolina	4	6,160	Copley et al. 1944[a]
Annual burn	Woodland	Tyler, Texas	100	720	Pope et al. 1946[a]
Prescribed fire	Shortleaf, loblolly pine	East Texas	200	420	Ferguson 1957[a]
Wildfire	Chaparral	California	4,940	49,360	Krammes 1960[b]
Wildfire	Chaparral	Central Arizona	156	184,040	Glendening et al. 1961[b]
Understory burn	Ponderosa pine	California	0[c]	0	Biswell and Schultz 1957[b]
Prescribed burn	Bluestem grasses	Upper coastal plain, northern Mississippi	74 to 84	1,542 to 5,759	Ursic 1969
Prescribed underburn	Post oak, hickories, blackjack oak	Upper coastal plain, northern Mississippi	148 to 300	868 to 1,179	Ursic 1970
Clearcut, slash burned	Western larch Douglas-fir	Western Montana	0[d] 0[e]	150 134	DeByle and Packer 1972[a]

Prescribed fire, pile and burn	Juniper	Central Texas			Wright et al. 1976
		3 to 4% slope	18	18	
		8 to 20% slope	146	504	
		37 to 61% slope	6	5554	
Wildfire	Ponderosa pine	Northwestern Arizona	3	1,254	Campbell et al. 1977
Wildfire	Ponderosa pine, Douglas-fir	Eastern Washington	12 to 35	146 to 2100	Helvey 1980 and Helvey et al. 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
Wildfire, clearcut	Douglas-fir tanoak, madrone	Southern Oregon		79[f g]	Amaranthus 1989
Wildfire, grass seeded plus fertilizer				56	
Clearcut and prescribed fire:	Oak spp., shortleaf pine	Northwestern South Carolina	Not known		Robichaud and Waldrop 1994
low severity				12.1	
high severity				502	

Wells et al. 1979.

[b] Cited in Tiedemann et al. 1979.

[c] No surface runoff and no erosion observed.

[d] Snowmelt.

[e] Summer storms.

[f] ns = not significant (p • 0.05).

[g] From Oct. 13 to May 4, after September fire.

Table 5.9. pH in water after fire only or fire in combination with other treatments usually remains fairly constant. EPA (1988) secondary standard specifies a pH between 6.5 and 8.5.

Pretreatment or Treatment	Habitat	Location	control	Posttreatment	Author(s) year
Wildfire Wildfire and N fertilization	Ponderosa pine, Douglas-fir	Eastern Washington		7.1 to 9.5 during first 8 mon. after fire; 9.2, 2 days after N fertilization	Tiedemann 1973
Wildfire	Mixed conifer, shrub	Central Sierra Nevada Mountains, California	~7.0 to 6.2[a]	~7.0 to 6.6[a]	Hoffman and Ferreira 1976
Pile, burn.	Juniper	Central Texas			
		3 to 4% slope	7.3	7.3	Wright et al. 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	6.22	6.1 to 6.3	Tarapchak and Wright 1977
Wildfire Wildfire and N fertilization	Ponderosa pine, Douglas-fir	Eastern Washington	7.4 to 7.6	7.4 to 7.6	Tiedemann et al. 1978
Prescribed fire	Ponderosa pine	Central Arizona	6.2	6.4	Sims et al. 1981
Pile, burn and seed	Juniper	Central Texas	7.1	7.3	Wright et al. 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 National Park, Wyoming	Yellowstone Lake	7.4	7.5	Lathrop, Jr. 1994

From May to July during the summer following the August fire.

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Table 5.10. Nitrate-N concentration in water after fire only or fire in combination with other treatments remained well below the EPA (1988) primary standard of 10 ppm except in one study.

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Author(s) year
			- - - - - maximum NO_3-N, ppm - - - - -		
Clearcut, slash burned	Douglas-fir	Western Oregon	0.1	0.43	Fredriksen 1971
Clearcut, slash burned	Douglas-fir, red alder	Western Oregon	0.7	2.1	Brown et al. 1973[a]
Wildfire Wildfire and N fertilization	Ponderosa pine, Douglas-fir	Eastern Washington	0.016[b] 0.005	0.042 0.310[b]	Tiedemann 1973
Wildfire, maintained in grass cover	Chaparral	Central Arizona	0.1	1.9	Longstreth and Patten 1975[a]
Clearcut, slash broadcast burned	White pine, western hemlock, grand fir	Northern Idaho	0.8	7.6	Snyder et al 1975[a]
Wildfire	Mixed conifer, shrub	Central Sierra Nevada Mountains, California	~0.06[b c]	~0.12	Hoffman and Ferreira 1976
Wildfire	Ponderosa pine	Northwestern Arizona	0.086	0.212	Campbell et al. 1977
Wildfire	Pine, spruce, fir, aspen, birch[d]	Northeastern Minnesota lakes	0.17	0.08 to 0.17	Tarapchak and Wright 1977

Wildfire Wildfire and N Fertilization	Ponderosa pine, Douglas-fir	Eastern Washington	<0.016[b] <0.016[b]	0.56 0.54 to 1.47	Tiedemann et al. 1978
Prescribed fire		Lower coastal plain, South Carolina	pretreatment not significantly different from posttreatment	0.02	Richter et al. 1982
Prescribed underburn	Loblolly pine plantation	Upper Piedmont, South Carolina	0.05	0.05	Douglass and Van Lear 1983
Clearcut, slash broadcast burned	Douglas-fir, ponderosa pine	Southern Idaho	0.02	0.05	Clayton and Kennedy 1985
Clearcut (41% of watershed), slash piled and burned	Grand fir, Engelmann spruce, subalpine fir,	Northeastern Oregon	0.092	0.162[e]	Tiedemann et al. 1988
Clearcut (17% of watershed), slash piled and burned	western larch		0.004	0.026[f]	
Prescribed burn, moderate	Ponderosa pine, gambel oak	Central Arizona	0.0013[b]	0.0029	Gottfried and DeBano 1990
Wildfire	Chaparral	Lexington Reservoir, Santa Clara Co. Calif.	0.02	0.04	Taylor et al. 1993
Prescribed underburn	White fir, giant sequoia, red fir, sugar pine, and Jeffrey pine	Sequoia National Park, California	0.001 to 0.005	0.010 to 0.394	Chorover et al. 1994
Prescribed broadcast: moderate burn severe burn	Chaparral	Southern California	 2.5 2.5	 9.5 15.3	Riggan et al. 1994
Wildfire	Lodgepole pine, Douglas-fir,	Glacier National Park, Montana	<0.040	0.124 to 0.312	Hauer and Spencer 1998

	ponderosa pine, western larch	

[a] Cited in Tiedemann et al. 1979.
[b] Maximum level attained.
[c] Mean concentration from May to July after August fire.
[d] Cited in Wright and Watts 1969.
[e] Posttreatment mean is significantly different from pretreatment mean at $p < 0.05$.
[f] Posttreatment mean is significantly different from pretreatment mean at $p < 0.01$.

Table 5.11. Sulfate concentration in water after fire only or fire in combination with other treatments did not approach the EPA (1988) secondary standard of 250 ppm.

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Author(s) year
			- - - - -Sulfate, ppm	- - - - -	
Wildfire	Taiga	Interior Alaska	7.12 to 66	8.3 to 80.7	Lotspeich et al. 1970
Wildfire	Mixed conifer, shrub	Central Sierra Nevada Mountains, California	1.5	1.7	Hoffman and Ferreira 1976
Wildfire	Pine, spruce, fir, aspen, birch[a]	Northeastern Minnesota lakes	1.17	1.79 to 1.86	Tarapchak and Wright 1977
Prescribed underburn	White fir, giant sequoia, red fir,	Sequoia National Park, California sugar pine, and Jeffrey pine			Chorover et al. 1994
Yellowstone wildfires	Subalpine lakes, Yellowstone National	Yellowstone Lake Lewis Lake Park, Wyoming	65.8[e] 70	64.8 76	Lathrop, Jr. 1994

[a] Cited in Wright and Watts 1969.

Table 5.12. Chloride concentration in water after fire only or fire in combination with other treatments did not approach the EPA (1988) secondary standard of 250 ppm

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Author(s) year
			- - - - - Chloride, ppm - - - - -		
Wildfire	Taiga	Interior Alaska	0.9 to 5.0	1.2 to 4.6	Lotspeich et al. 1970
Wildfire	Mixed conifer, shrub	Central Sierra Nevada Mountains, California	0.6	1.0	Hoffman and Ferreira 1976
Wildfire	Pine, spruce, fir, aspen, birch[a]	Northeastern Minnesota lakes	0.80 to 0.89	1.24	Tarapchak and Wright 1977
Prescribed underburn	White fir, giant sequoia, red fir,	Sequoia National Park, California sugar pine, and Jeffrey pine	0.49 to 0.56	0.40 to 2.78	Chorover et al. 1994
Yellowstone wildfires	Subalpine lakes, Yellowstone National	Yellowstone Lake Lewis Lake Park, Wyoming	5.1 6.4	3.6 7.1	Lathrop, Jr. 1994

[a] Cited in Wright and Watts 1969.

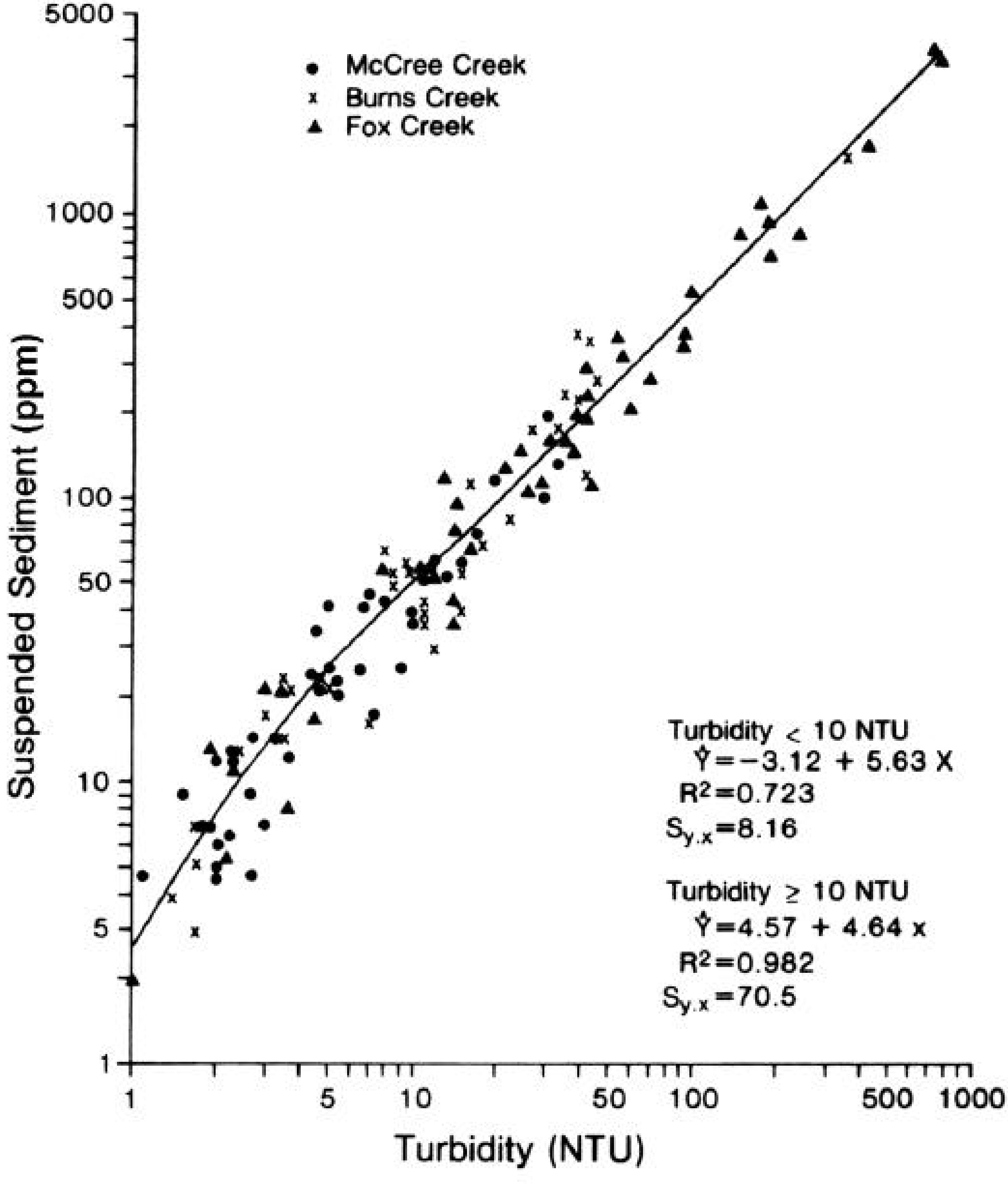

Figure 5.2 Relationship between turbidity (nephelometric turbidity units - NTUs) and suspended sediment (ppm) (Helvey et al. 1985).

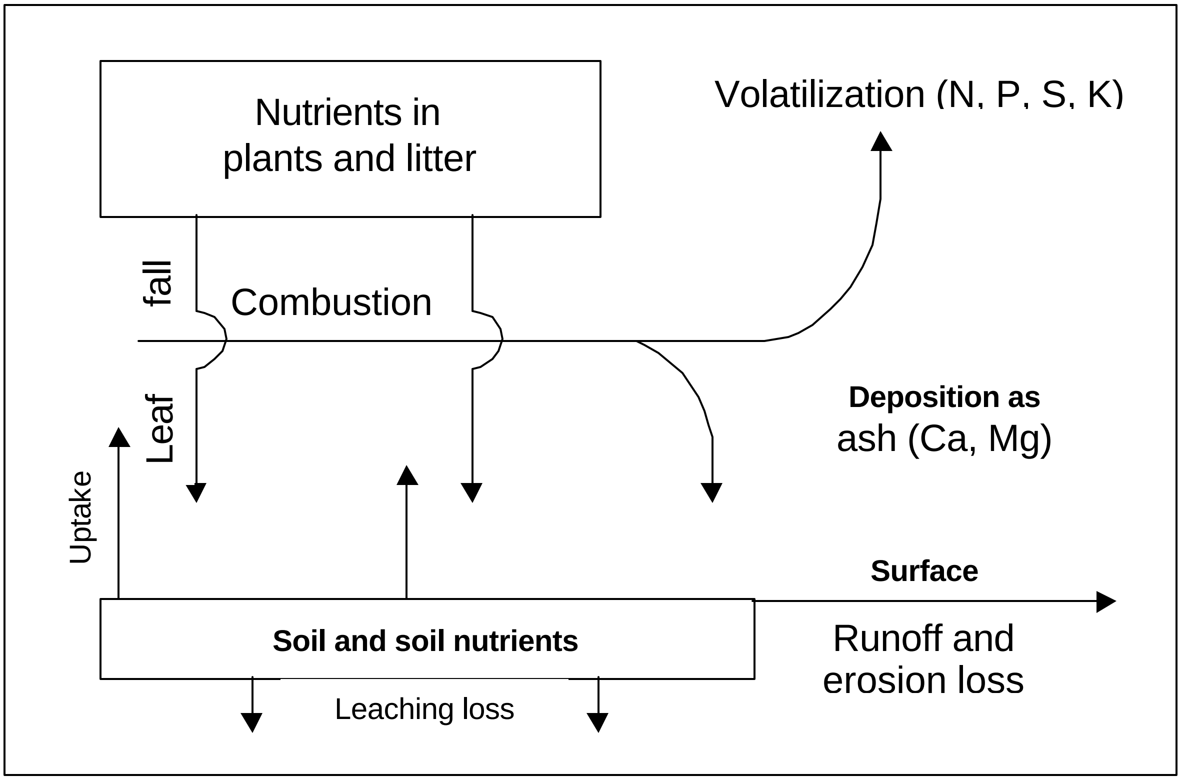

Figure 5.3. Possible pathways of plant- and litter-contained nutrients in response to combustion (Tiedemann 1981).

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Pesticides

J. L. Michael

Introduction - In managing 191.8 million acres of national forests (NFs), the U.S. Forest Service (USFS) practices vegetation management for a number of purposes, including protecting desirable vegetation from pathogens, competing vegetation, insects, and animals. Vegetation management is also practiced on road and utility rights-of-way, to improve recreation areas and wildlife habitat, and to control noxious weeds. The Forest Service requires training of personnel who recommend and use pesticides, applicator certification, and safety plans. Non-national forest lands are treated with pesticides for many of the same purposes, but often more intensely.

To meet the minimum requirements of the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) at the state level, the Environmental Protection Agency has established and maintains cooperative enforcement agreements in 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) groundwater protection.

When forestry pesticides are used near water on federal, state, or privately owned lands, buffer zones are left between the treatment and the water resource. The width of the buffer varies by site conditions, site sensitivity, and local or state recommendations. NFs in some states use more conservative buffers than those recommended by the state. However, a major research gap exists concerning the effectiveness and functioning of buffers. It is not possible to determine the minimum buffer width to protect streams from either pesticide or sediment contamination.

Issues and Risks **-** Less than 1% of National Forest System (NFS) lands receive some form of pesticide treatment annually. The amount of pesticide used and the number of acres treated varies slightly from year to year (Table 5.13). Biological control agents are also used, including Bacillus thuringiensis (Bt) and nucleopolyhedrosis virus (Npv) for western spruce budworm and gypsy moth control. Table 1 includes the acres treated with these two biological agents, but does not include Bt or Npv in the active ingredient column because the unit of measure is different than used for other pesticides. NF lands were treated with approximately 272,577 pounds of Bt in FY97, which is about average unless there is an unusual outbreak of budworm or gypsy moth. The average acreage treated is approximately 30,000 acres per year FYs 1990-1997, excluding 1992 when there was a large outbreak of these insects.

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. Control of pathogens is accomplished by using fungicides and fumigants, including treatment of seeds, seedling nurseries, greenhouses, seedtree nurseries, and individual trees. In 1997, the USFS treated 35 greenhouses with a total of 11.4 pounds active ingredient (ai) of fungicides and fumigants including benomyl, chlorothalonil, dicloran, iprodione, metalaxyl, propiconazole, thiophanate methyl, and triadimefon. Dicloran accounted for 9.1 lbs of this use. However, most fungicide and fumigant use occurs on small acreages in nurseries for disease control (Table 5.14). Of the fungicides and fumigants used for disease control on NF land, 55.4% of the active ingredient (including chloropicrin, dazomet, and methyl bromide) was applied to small parcels of land which represented only 1.1% of the acres treated in 1997 (Table 5.14). The remaining 44.6% was spread over 98.9% of the treated land.

Table 5.13. Total pesticide use on National Forest land since 1989.

Fiscal Year	Acres Treated	Pounds Active Ingredient*
1990	333,587	266,789
1991	310,454	269,674
1992	598,125	274,427
1993	288,977	213,727
1994	256,778	184,786
1995	306,009	189,276
1996	220,290	173,561
1997	297,880	200,841

* Does not include biological control agents.
From: USFS 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998.

Vegetation management is frequently taken to mean the control of competing vegetation in timber management programs. On NFS lands, more area was treated to protect vegetation from animals (25.9% of all treated land) and insects (22.3%), and to control noxious weeds (19.5%) than for control of competing vegetation (17.9%) in FY97.

Animal damage control used 112 pounds of strychnine spread over 48,655 acres and 5,399 pounds of putrescent egg solids spread over 15,644 acres. While very small amounts of strychnine were used over vast acreages, it is very toxic (Table 3). Putrescent egg solids, however, are derived from food products and the USEPA has waived toxicology requirements. These two products account for more than 96% of the active ingredients used in protection of vegetation from animals. Insect control relied mainly on biological agents, but some insecticides and oils are used. The insecticides included carbaryl (805 lbs on 165 acres), and chlorpyrifos (73 lbs on 67 acres). Dormant oil (160 lbs) was used on 16 acres for control of a variety of insects and their eggs. These three represent 94.6% of all chemical insecticides used on NFS lands.

Competing vegetation control is by herbicides, algicides, and plant growth regulators. Management objectives for the herbicide use and the NF acres treated in FY97 are listed in Table 5.14. Many acres are treated for timber management, i.e. planting site preparation and for 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 years, or longer than 150. Thus, herbicides used only once or twice in 20 or more years. The total NFS land area treated for timber production in FY97 was approximately 53,283 acres, including site preparation, conifer release, and hardwood release (Table 5.14).

Table 5.14. Vegetation management objectives for pesticides used on USFS lands in FY97. Data extracted from Table 10 of the Annual Report of the Forest Service (USFS 1998).

Pesticide Type Management Objective	Acres Treated	% of Treated Acres	Pounds AI Applied	% of Applied AI
Fungicides and Fumigants				
General Disease Control	31758.60	98.436	16476.77	21.235
Nursery Disease Control	362.32	1.123	42980.15	55.392
Fungus Control	107.23	0.332	5615.73	7.238
Soil Fumigation	35.20	0.109	12519.50	16.135
Total	32263.35	100.000	77592.15	100.000
Herbicides, Algicides, Plant Growth Regulators				
Noxious Weed Control	57973.37	48.752	35144.75	31.228
Planting Site Preparation	29870.00	25.119	43205.44	38.390
Conifer Release	21292.09	17.905	24069.75	21.387
Wildlife Habitat	2307.30	1.940	2579.81	2.292
Hardwood Release	1465.00	1.232	1049.30	0.932
Nursery Weed and Disease Control	1013.70	0.852	1384.07	1.230
Rights-of-Way	859.90	0.723	2438.88	2.167
Hardwood Control	662.00	0.557	387.40	0.344
Seed Orchard Protection	180.80	0.152	139.18	0.124
Recreation Improvement	172.00	0.145	512.10	0.455
Aquatic Vegetation Control	6.00	0.005	2.00	0.002
Total	118908.31	99.999	112540.11	99.999
Acaricides, Insecticides, Pheromones, Predacides, Repellents, Rodenticides				
Animal Damage Control	77253.24	52.904	5724.79	2.025
Insect Control-Biological	65784.00	45.050	272841.41	96.511
Insect Control- Chemical	780.19	0.534	1096.98	0.388
Vector/Plague Suppression	1714	1.174	52.47	0.018
Seed Orchard Protection	409.24	0.280	4.73	0.002
Recreation Improvement	65	0.044	2982	1.055
Fish Eradication	20	0.014	3.12	0.001
Total	146025.67	100.000	282705.50	100.000

Noxious weed control accounted for 48.8 percent of the area treated by herbicides (table 5.14) or 19.5 percent of the vegetation management area. Noxious weeds are usually non-native plants that, lacking natural controls, spread quickly taking over or ruining habitat for native plants. They generally possess one or more of the following characteristics: aggressive and difficult to manage, poisonous, toxic, parasitic, a carrier or host of serious insects or disease. There are 74 terrestrial species on the federal noxious weed list, including the most recently addition, *Pueraria lobeta* (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 and noxious weed control accounted for 94 percent of all acres treated with herbicides.

Protecting forests and seedlings is the single largest component of the vegetation management program. 48.2 percent of all NFS treated lands in FY97 were treated for protection of vegetation from damage by animals (25.9%, mostly rabbits and deer) and insects (22.3%, mostly western spruce budworm and gypsy moth).

One major issue with pesticides use in the vegetation management is their impacts on drinking water quality. Of the 75 pesticides used alone and in combination, 20 account for more than 95% of the total active ingredients applied to NFS lands in FY97, excluding the biological control agents, Bt and Npv. To adversely impact drinking water, they must 1) be toxic to humans, and 2) reach drinking water sources at concentrations exceeding drinking water standards.

Toxicity

Findings - 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% 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 USEPA 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 from 10 up depending on EPA's evaluation of the reliability of the test data.

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 cause adverse effects (USEPA 1993). NOAEL is derived from the test data where all doses have some effect, but some of the observed effects are not considered adverse health effects. 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 lbs. and consumes 2 pints of water per day, and a child weighs 22 lbs. and consumes 1 pint of water per day over the period of exposure. HALs are calculated for one-day, ten-day, longer-term (90 to 365 days), or lifetime (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 (USEPA 1993). The safety factor can range from 1, but rarely less than 10, to 10,000 depending on the available 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 NOAELs 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 identifying a LOAEL but not a NOAEL for one or more animal species are available, a safety factor of 1,000 is used. For situations where good chronic data are absent, but subchronic data identify a LOAEL but not a NOAEL, the safety factor of 10,000 is used. EPA's estimates of safe levels for daily exposure to the most widely used

Table 5.15. Estimates of safe levels for daily exposure to the 20 pesticides most used on NFS lands in FY97 in the vegetation management program.

Pesticide	Amount Used	All Pesticides	RfD	NOEL	NOAEL	Lifetime HAL
	Pounds	Percent	mg/kg	mg/kg	mg/kg	mg/L
Dazomet	32519	16.19	NA[1]			
Hexazinone	32225	16.05	0.033	25		0.200
Glyphosate	25114	12.50	0.1	20		0.700
2,4-D	17864	8.89	0.01		1	0.070
Borax	16466	8.20	0.09		8.8	0.60[2]
Chloropicrin	14415	7.18	NA[1]			
Picloram	13983	6.96	0.007	7		0.500
Methyl bromide	13480	6.71	0.0014		1.4	0.010
Triclopyr	13450	6.70	0.25	25		NA
Putrescent egg Solids	5399	2.69	NA[3]			
Imazapyr	3804	1.89	NA	300		
Carbaryl	805	0.40	0.1	NA	9.6	0.700
Dicamba	525	0.26	0.03	NA	3	0.200
Clopyralid	297	0.15	NA	NA	NA	NA
Dormant oil	160	0.08	NA	NA	NA	NA
Strychnine	112	0.06	0.0003		NONE	NA
Zinc phosphide	104	0.05	0.0003		NONE	NA
Thiram	100	0.05	0.005	5		NA
Chlorpyrifos	73	0.04	0.003	0.03		0.020
Metsulfuron	38	0.02	0.25	25		NA

NA Not available.

[1] These fumigants are not expected to get into water.

[2] HAL for elemental boron. From USEPA, Office of Drinking Water, November 1990.

[3] Made from food products, toxicology was waived by EPA.

pesticides on NFS lands are summarized in Table 5.15. Of the pesticides listed in Table 5.15, only elemental boron (potentially from borax) and methyl bromide are listed in EPA's drinking water contaminant candidate list for consideration for possible regulation.

Findings: Water Contamination - The pesticides used by 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 (less than 30 ft), usually in concentrations far below USEPA drinking water standards and the occurrence is infrequent. Table 5.16 summarizes reports of pesticides from Table 3 that have been detected in surface waters in the US.

Larson et al. (1997) summarized the results of 236 studies throughout the U.S.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. They reported monitoring results for 52 pesticides approved for agricultural, urban and forestry use and their metabolic byproducts. In Table 5.15, only 6 of the pesticides listed were reported to be present in surface water by Larson et al. (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 these concentrations exceeded USEPA human health standards and most were less than 0.002 mg/L. It is important to recognize that surface water is not necessarily drinking water. The studies summarized by Larson et al. (1997) dealt with surface water, principally lake, reservoir, and river water, which would be treated prior to use for drinking water.

Reports of pesticide contamination of water are usually from agricultural (Kolpin et al. 1995; Koterba et al. 1993) or urban applications (Bruce and McMahon 1996), but the potential for contamination from forest vegetation management programs exists. When forest sites are compared with other land uses, water from forests is generally much less contaminated. Several studies on forest sites listed in Table 5.16 present data for water collected directly from treated areas and the concentration of pesticides can appear quite high compared to samples taken from large rivers and lakes. Pesticide concentrations become significantly diluted as they flow from the treated sites to downstream locations.

Cavalier et al (1989) monitored 119 wells, springs, and municipal water supplies from 1985-1987 for occurrence of pesticides in drinking water throughout the state of Arkansas. They did not detect drinking water contamination from the forestry pesticides for which they monitored (2,4-D, hexazinone, and picloram).

Michael and Neary (1993) reported on 23 studies conducted on industrial forest lands 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 (less than 300 acres) and the streams would not be sufficient to be public drinking water source, but their flow reached downstream reservoirs. The maximum observed hexazinone, imazapyr, picloram and sulfometuron concentrations in streams on these treated sites did not exceed drinking water standards (HALs), except for one case in which hexazinone was experimentally applied directly to the stream channel. Even in this case, 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 drinking water standards during the year after application.

Bush et.al. (1990) reported on use of hexazinone on 2 coastal plain sites (deep sand and sandy loam soils) with monitoring for impacts on groundwater. Hexazinone was not detected in groundwater at the South Carolina site for 2 years after application. In Florida, hexazinone was found infrequently in shallow test wells at concentrations up to .035 mg/L, much lower than the established drinking water standard (Table 5.15). Water from these sites drains into other creeks and rivers, and is diluted before entering reservoirs.

Michael et al (1999) reported the dilution of hexazinone downstream of treated sites. One mile below the treated site, hexazinone concentrations were diluted 3-5 times below the maximum concentration observed on the treated site. Hexazinone was applied for site preparation at 6 lb ai/a to clay loam soils, a rate 3 times the normal, and it was applied directly to a stream segment, resulting in a maximum observed on-site concentration of 0.473 mg/L. This was more than twice the lifetime HAL but considerably below the longer-term HAL (10% of life expectancy or about 7 years) of 9.0 mg/L (USEPA 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. 166 acres of a 603 acre watershed were aerially sprayed with 1 lb ai/a of dicamba. A small stream segment was also sprayed resulting in detectable dicamba residues 2 hours after application began, approximately 0.8 miles down stream. Concentrations rose for approximately 5.2 hrs after treatment began and reached a maximum concentration of 0.037 mg/L, less than a fifth of the HAL (Table5.15). No dicamba residues were detected beyond 11 days after treatment.

2,4-D and glyphosate have aquatic labels, which permit direct application to water. Stanley et.al. (1974) found that when 2,4-D was applied for aquatic weed control, about half of water samples from within treatment areas contained 2,4-D, and the highest concentration (0.027 mg/L) was less than half of the drinking water standard. Newton et al. (1994) aerially applied glyphosate at 3 times the normal forestry usage rate (4 lbs ai/a), no buffers were left, and all streams and ponds were sprayed. Initial water concentrations were 0.031 and 0.035 mg/L in Oregon and Georgia and 1.237 mg/L in Michigan on the day of application. After Day 1, glyphosate concentrations dropped to below 0.008 mg/L on all three sites for the duration of the study. Drinking water standards were 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 et al (1991) found trace residues of metsulfuron in shallow monitoring wells in Florida. 24 wells were sampled to a depth of 6 feet. Metsulfuron was detected (0.002 mg/L) in one of 207 samples collected over 2 months following application.

There are several pesticides in Table 5.15, which have not been reported in water. They include chloropicrin, chlopyralid, dazomet, and thiram. Chloropicrin and dazomet are soil fumigants and are used only for seedling production. Chlopyralid is a relatively new compound in the US. 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 lands are used in nurseries. Such nursery-intense use may result in localized groundwater 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 fumigants for fumigating seedbeds. Dazomet is a soil fumigant, which is relatively insoluble in water (3 g/L). However, dazomet is unstable in water and quickly breaks down into methyl isothiocyanate (MITC), formaldehyde, monomethylamine, and hydrogen sulfide. All three are toxic, but the most toxic is MITC. The RfD for formaldehyde is 0.2 mg/kg/d. However, EPA has classified formaldehyde as a of medium carcinogenic hazard to humans. Methyl bromide is very toxic. There is insufficient data to determine whether frequent use of these three pesticides adversely impacts water quality, either locally or over an expanded area.

Reliability of Findings – Most data reviewed in this section come from scientific literature. The data listed in Table 5.16 and derived from Larson et al. (1997) were extracted from in-house reports from the US Geological Survey, the US Environmental Protection Agency, and state and local governmental departments for the environment. The remaining reports are published in open scientific literature and are the most reliable data, because they were subject to peer review and have been exposed to scrutiny with respect to validity of methods, completeness of data and interpretation of the data. Monitoring data from in-house publications and reports may be less reliable.

Limitations of the data are obvious for the few chemicals that have not been investigated. Chemicals with little information could benefit from additional data. Additional limitations include lack of sufficient testing for health effects as indicated in Table 5.15. The question of cumulative toxicological effects has not been addressed for any of the tank mixtures utilized in modern forest management. Indeed it would be almost impossible to conduct such studies because there are so many combinations and environmental interactions possible.

Data on pesticide contamination of water for the pesticides in Table 3 have been collected from a number of locations around the world and the findings are generally similar. Use of most pesticides on NFS lands can not be expected to contribute significantly to groundwater or drinking water contamination, because of their infrequent use.

Ability to Extrapolate Findings - Care must always be exercised in extrapolating data from local studies on drinking water to a regional or larger scale. However, three strategies 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. These studies have, with a few exceptions, generally confirmed the lack of unacceptable impacts on drinking water safety. The exceptions included those cases in which pesticide was applied directly to water and the high concentrations lasted on a few hours.

Table 5.16. Frequency and occurrence of water contamination from pesticide use in North America. This list is not comprehensive, but provides a range of values which is representative of the literature from around the world.

Pesticide	Location	Maximum (mg/L)	Range (mg/L)	Comments	Reference
2,4-D	Large River Basins Throughout US	0.0075	0.00004-0.0075[1]	24 reports of mainly urban, sub-urban, agricultural sources	In: Larson et al. 1997
	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
	Saskatchewan, Can	0.0000007	ng	Natural spring flow	Wood & Anthony 1997
Borax	nr	nr	nr	nr	nr
Carbaryl	Mississippi River	0.0001	ng	1 report	In: Larson et al. 1997
	New Brunswick, Can	0.314	ng	Aerial spray spruce budworm control	Sundaram & Szeto 1987
	New Brunswick, Can	0.314	0.123-0.314	Budworm control	Holmes et al. 1981
	Montana	0.260	ng	Western spruce budworm control	Pieper & Roberts 1978
Chloropicrin	nr	nr	nr	nr	nr
Chlorpyrifos	Mississippi River, the Lower Colorado River, rivers and lakes in Kansas, and irrigation ditches in California,	0.00015	0.00004-0.00015	3 reports	In: Larson et al. 1997

	Arizona, Nevada				
Clopyralid	nr	nr	nr	nr	nr
Dazomet	nr	nr	nr	nr	nr
Dicamba	USFS land near Hebo, OR	0.037	0.006-0.037	Treated 166 ac of 603 ac forest catchment. Highest concentration diluted to 0.006 mg/L 2.2 miles downstream.	Norris 1975
Glyphosate	45 ha Coastal British Columbian catchment	0.162	0.0032-0.162	Highest concentration in streams intentionally sprayed, lowest in streams with smz	Feng et al. 1990
	Quebec	3.080	0.078 to 3.08	9 of 36 streams contained glyphosate after forest spraying	Leveille et al 1993
	Ohio	5.2	ng	no-tillage establishment of fescue	Edwards et al. 1980
	Georgia Michigan Oregon	0.035 1.237 0.031	ng ng ng	Forest sites for scrub hardwood control and direct spray of streams	Newton et al. 1994
Hexazinone	Mississippi River	0.00007	ng	detected in 5 tributaries	In: Larson et al. 1997
	Alabama, Florida, Georgia	0.037	0.0013-0.037	7 reports, each treated catchment containing ephemeral/first order streams	Michael & Neary 1993
	Alabama	2.400	ng	Applied directly to ephemeral channel and in first runoff water	Miller & Bace 1980
	Alabama	0.473	0.422-0.473	Ephemeral/first order stream in catchments	Michael et al 1999

				treated with 3X rate of hexazinone in liquid and pellet formulation with accidental application to streams	
	Arkansas	0.014	ng	11.5 ha watershed drained by ephemeral to first order stream	Bouchard et al. 1985
	Georgia	0.442	ng	Ephemeral/first order stream in treated catchment, pellets applied to stream channel	Neary et al. 1986
Imazapyr	Alabama	0.680	0.130-0.680	2 reports, each treated catchment containing ephemeral/first order streams, herbicide accidentally applied to stream channel	Michael & Neary 1993
Methyl Bromide	nr	nr	nr	nr	nr
Metsulfuron	Central Florida	0.008	ng	Water in surface depression in slash pine site	Michael et al 1991
Picloram	North-central Arizona	0.32	ng	Pinyon-juniper site	Johnsen 1980
	Streams and rivers in N. Dakota, Wyoming, and Montana	0.005	0.00001-0.005	4 reports from mainly range-land uses	In: Larson et al. 1997
	Alabama	0.442	ng	Pellets accidentally applied directly	Michael et al. 1989

				to forest stream	
	Georgia, Kentucky, Tennessee	0.021	nd-0.021	6 study catchments with ephemeral/first order stream in each treated forest catchment	Michael & Neary 1993
	North Carolina	0.01	ng	ephemeral/first order stream in treated forest catchment	Neary et al 1985
	Saskatchewan, Can	0.000225		Natural spring flow	Wood & Anthony 1997
Strychnine	nr	nr	nr	nr	nr
Thiram	nr	nr	nr	nr	nr
Triclopyr	Florida	0.002	ng	Coastal plain flatwoods catchments near Gainesville, FL	Bush et al. 1988
	Ontario	0.35	0.23-0.35	Intentional aerial application to boreal forest stream	Thompson et al. 1991
Zinc phosphide	nr	nr	nr	nr	nr

[1] Range of maximum values reported as summarized by Larson et al. 1997
ng-not given, nr-not given

Some variability in results is due to regional soil and climate differences. In the South, infiltration rates on 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 site prepared land and may lead to much higher pesticide concentrations in stormflow than in other areas of the country with much higher infiltrations rates. Very high infiltration rates are typical of soils in the Pacific Northwest. There, if streams are protected by buffers, pesticides broadcast applied generally reach streams either via direct application or through baseflow contributions. The result is generally lower levels of contamination where infiltration rates are highest.

Secondary Linkages - Related issue is the protection of the aquatic ecosystem. Impacts of pesticides on aquatic ecosystem form, composition, and functioning will be related to the type, quantity, toxicity, and timing of pesticide introduction into surface waters.

Research Needs - Clearly two issues related to the vegetation management program emerge which need additional research. The first is the impacts of frequent, repeated use of fungicides and fumigants in nursery operations on nearby water quality. The second issue is buffer width and composition. There is too little information on the functional processes which permit buffers to mediate against pesticide contamination streams and surface waters in general. These processes must be identified and understood so managers can design and install optimally functional buffers ffers to protect the water resource and its associated aquatic ecosystem.

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Chapter 6

Recreation

Dispersed and concentrated recreation impact water quality and are discussed in this chapter.

Dispersed Recreation and Drinking Water Quality

David Cole

Issues - Dispersed recreation is a common use of National Forest (NF) lands with the potential for significant water quality impacts. Trails are constructed to provide access for recreationists. Runoff from trails can add sediment to streams, particularly at trail fords. Recreationists picnic, camp and walk off-trail and, in some places, off-road vehicles travel cross-country. The resultant loss of vegetation and soil compaction can lead to increased runoff, erosion and sedimentation.

Recreationists, their pets and recreational stock contaminate water supplies by carrying and depositing microorganisms that cause human diseases. Contamination comes from fecal deposition and from direct contact with water, through activities such as swimming and washing. Dispersed recreation on NFs has constantly increased and will certainly continue to increase in the future. Recreational behaviors are commonly unrestricted; visitor education is typically inadequate; and few facilities are provided to ensure proper disposal of human waste. Consequently, drinking water quality problems associated with recreation use are expected to increase in the future. In a recent survey of Forest Service watershed managers, recreation was the most commonly reported water quality concern. 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.

Findings - The impacts of dispersed recreation on sediment have not been quantified. While it is clear that recreation facilities (particularly trails) and recreation use elevate sediment levels, logic suggests that dispersed non-motorized recreation should rarely have a significant impact. Non-motorized recreation use simply does not disturb much of the watershed. Cole (1981) estimated that less than 0.5 percent of a heavily used portion of the Eagle Cap Wilderness, Oregon, was directly affected by trails and camping. Must of the disturbed area was located far enough from streams for the effect to be negligible. Recent research indicates that sediment yield from trails is much higher when trails are used by horses than used by hikers or llamas (DeLuca *et al.* 1998).

Impacts of dispersed motorized recreation (*e.g.* off-road vehicles) 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 non-motorized 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 (Wilshire *et al.* 1978).

It is a well-established fact, dispersed recreation can introduce pathogenic organisms into watersheds used as sources of drinking water. LeChevallier *et. al.* (1991), in a broad survey of surface municipal drinking water sources, found oocysts of *Cryptosporidium* and cysts of *Giardia* species even in protected watersheds. Suk *et al.* (1987) found cysts of *Giardia* in 27 of 78 samples from backcountry streams within several of the large wilderness areas in the Sierra Nevada mountains in California. Taylor *et al*. (1983) found *Campylobacter jejuni* and *Giardia lamblia* in the stools of 23% and 8% of persons reporting diarrhea, and in the streams in Grand Teton National Park, Wyoming. There is some controversy about whether mammals other than humans can spread these pathogenic organisms to humans. The question is: do horses, mules and dogs that accompany recreationists contribute to the contamination of surface waters? Since these animals are more likely to defecate directly in or near water, they may be a concern if they are important carriers. The issue of transmission by wild mammals is also relevant to the question of whether or not water quality can be adequately protected by eliminating or severely restricting recreation use.

Taylor *et al.* (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). Along with the finding that *C. parvum* readily crosses host species barriers, this has convinced most experts that human infections are often the result of transmission from wild animals 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” (p. 271). *Giardia* has been found in humans, beaver, muskrat, mule deer, sheep, cattle, elk, coyotes, dogs, cats, horses, moose and small rodents. It has been successfully cross-transmitted between humans and a number of these animals (Hibler and Hancock 1990).

These results suggest that wild animals, recreational pack animals and pets contribute pathogenic organisms to water supplies, although there are still questions about which pathogens are transmitted and the relative importance of humans and other mammals 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. In the absence of recreation, wild animals will still introduce and spread pathogenic microorganisms. Even with proper human waste disposal, pets and packstock will defecate in or near waters and spread pathogens.

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 *et al.* 1987, Varness *et al.* 1978), others report no correlation (*e.g.* Silsbee and Larson 1982, Auckerman and Springer 1976, Silverman and Erman 1979, Skinner *et al.* 1974, Gray and Adams 1985, Gray 1982) and at least one series of studies reports a negative correlation (Walter and Bottman 1967, Stuart *et al.* 1971). 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 positive correlations between recreational use and contamination, and between fecal coliform and the occurrence of *Giardia* and *Cryptosporidium* increase in strength as levels of contamination increase (LeChevallier *et al*. 1991).

The study finding a negative correlation between 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 *et al.* 1971: 1048).

From these findings, several implications can be drawn. First, surface waters are likely to be unsafe for drinking even where recreation use is excluded. In fact, Suk *et al* (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. Backcountry 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)*.

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 heavy dispersed recreation use areas include reducing recreation use, prohibiting pack animals and pets, providing adequate toilet facilities, and educating recreationists in appropriate waste disposal techniques (see for example Meyer 1994, Hampton and Cole 1995).

The relationship between amount of dispersed recreation and water contamination depends on other variables, including the type of recreation use, soils, slope, climate, and so on. None of these relationships have 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 recreationists in the proper disposal of human waste is suggested by studies of the survival of feces buried in soil in Montana. Samples of feces were inoculated with two bacteria, *Esherichia coli* and *Salmonella typhimurium* and both survived in large numbers for 8 weeks after burial in early summer (Temple *et al.* 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 *et al.* 1982). Clearly, the idea that shallow burial (in "cat-holes") 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.

Logic suggests that, in dispersed recreation areas, fecal deposition is a much more substantial source of water contamination than water contact activities, such as swimming and wading. However, there are no data available to test this contention.

Reliability, Confidence, and Limitations - 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.

Ability to Extrapolate Findings - 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, recreationists are particularly drawn to water sources, increasing the likelihood of contamination. Moreover, in arid lands, water systems are not "flushed" as rapidly or frequently as in places that receive more precipitation, making contamination problems more severe. Flack *et al.* (1988) suggest that periods of low flow may exacerbate the effect of intense recreation use on water quality.

Ability to Address Issues - These findings suggest some obvious management approaches for dealing with dispersed recreation and drinking water. Since dispersed recreation can contribute to contamination, every affordable effort should be made to educate recreationists in appropriate human waste disposal and to provide well-designed and appropriately located facilities for the disposal of human waste. They should be informed of the need to purify all drinking water taken from surface water sources, given the potential for contamination.

Where recreation use is high and water contamination is too high, sanitary facilities need to be developed or improved, and/or use restricted. Where it is clear that dispersed recreation use levels are low, use restrictions and the provision of sanitary facilities are not worth the costs involved. In the middle ground, our level of understanding is inadequate to suggest whether it is worth the costs of limiting access, restricting behavior or investing in sanitary facilities.

Research Needs **–** 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, though there are few cases where virus isolations from drinking 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 *Cryptosporidium parvum* and *Giardia lamblia* is particularly important.

Additional research is needed to provide a more solid foundation for decisions about where and how to restrict dispersed recreation use 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 human recreationists, their pets and pack animals. Thresholds of use need to be identified, above which adverse effects on water quality become pronounced and become 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.

Research needs to develop techniques to distinguish between human and other sources of contamination, and to assess the validity of rules of thumb managers use to develop management prescriptions and the effectiveness of techniques managers develop to mitigate contamination.

More research is needed on fecal decomposition rates, on variables that influence decomposition rates, and on how pathogens disperse within and across 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.

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Effects Of Concentrated Recreation on Drinking Water Supplies

Myriam Ibarra and Wayne C. Zipperer

Introduction - Lands with natural cover are one of the most important sources of public recreation. Increased demands for outdoor recreation result in greater need of drinking water and in increased amounts of wastewater. At the same time that expanding recreational resorts invite larger number of visitors, private lands adjacent to forests are magnets for real-estate development. This development may negatively affect drinking water and alter hydrologic processes.

Drinking water supply facilities in federal, state, local and private forests and grasslands are generally small and located in pristine environments. Small municipality systems are more likely to have impaired waters than the larger ones because they do not benefit from strict water monitoring and treatment. To illustrate the effects of concentrated recreation on drinking water supplies, we use data from the National Forests; however, our findings and recommendations are applicable to managers of other public lands.

The USDA Forest Service, with its National Forest and Grassland system, is the single largest supplier of public outdoor recreation in the United States. The National Forests offer visitors 4,385 miles of the National Wild and Scenic River System, 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 US Forest Service hosted more than 800 million recreational visits that included skiing, hiking, camping, boating, fishing, hunting, and pleasure driving. That number is expected to grow to 1.2 billion by 2050. Recreation revenues exceeded $45.2 million to the US treasury in 1997 (USDA Forest Service 1998b).

The Forest Service owns and 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 recreational sites and facilities. The Forest Service manages all water systems as public systems in accordance with Environmental Protection Agency (EPA) and respective state regulations. In many cases, this approach exceeds minimum requirements for system operation (USDA Forest Service 1981).

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 waters supplies, while the greater impact of concentrated winter recreation's may be in ground water. This chapter deals with the effects of increased vehicular traffic, water recreation, and winter recreation. The effects of visitor centers and development are discussed in more detail in Chapter 7.

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 and migrate to surface and ground waters 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%, and the number of visits was highly correlated with the number of vehicles (Cordell et al. 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. Based mainly on tourism to ski resorts in Colorado's National Forests, construction plans for the zone between Denver and its international airport are projected to double the populations of several towns (Brooke 1999). Improvement and expansion of parking lots and roads increase peak runoff and non-point source pollution from impervious surfaces. Runoff may be contaminated with salt, heavy metals, petroleum residues or landscaping chemicals, and potentially degrade surface and ground water quality (U. S. Environmental Protection Agency 1983). Oxygenates and polycyclic aromatic hydrocarbons (PAHs) are gasoline residues that have been found in drinking water supplies, and are potential threats to human health.

Findings -. In an attempt to curb air pollution, EPA mandated the use of Reformulated Gasoline (RFG) (U. S. Environmental Protection Agency 1994). RFG differs from conventional gasoline in that it has reduced vapor pressure, reduced sulfur content, and reduced aromatic and benzene content. RFG burns more completely, thereby decreasing damaging engine deposits. RFG may or may not be blended with "oxygenates", an additives that contain oxygen. Oxygenates have been used as fuel additives in gas since the late seventies, mostly in premium gasoline to improve the octane ratings. The most widely used oxygenate (MTBE - methyl tertiary butyl ether) had been tested rigorously in laboratories and determined to be safe for use in gasoline. MTBE was expected to help fuels burn cleaner and more completely, thereby producing 15 - 17% less pollution than conventional gasoline (Keller et al. 1998). The use of oxygenates in gasoline during three years did not produce statistically significant air quality benefits, but was found to contaminate local water sources.

MTBE is highly soluble in water and transfers readily to ground water from leaky underground storage tanks, pipelines and other components of the gasoline distribution system. Also, vehicular emissions of oxygenate dissolve in rainfall, and subsequently, infiltrate to the groundwater and may result in widespread impacts at low concentration (Allen et al. 1998). The latter is the kind of contamination that could occur in very popular National Forest resorts during the peak visitor season.

Even though the carcinogenic properties of MTBE and PAH's are yet to be fully understood, the US EPA states that they are animal carcinogens and may poses some human cancer risk. The California EPA proposed a maximum of 14 ppb of MTBE in drinking water to protect the people against the risk of cancers (Keller et al. 1998).

EPA has set a drinking water advisory for MTBE at 20-40 μg/L, based on consumer acceptance of taste and smell. When highly concentrated, MTBE smells like turpentine and tastes like paint thinner. A small amount can make water distasteful and undrinkable. Urban centers in California have monitoring programs for drinking water that shut down a drinking water supply when the concentration of MTBE is equal or greater than 5 μg/L. If that is the case, alternative water sources are used. In contrast, because of the inadequacy of small public systems or private drinking water supplies to monitor water quality, drinking water may contain concentrations higher than 5 μg/L. Proposed legislation would direct the EPA to make cleaning up storage tanks that leak gasoline into drinking water a top priority.

Water Recreation

Issues and Risks - Concentrated recreation on surface waters produces chemical and microbial contamination of surface drinking water sources. Individual boats, marinas and swimmers usually release only small amounts of pollutants that can go undetected. Yet when multiplied by thousands, 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, Gelt 1998). The effects of swimmers on drinking water supplies is an emerging problem that has prompted some utilities to limit or ban recreational uses in their reservoirs. Persons 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 MTBE in motor boats, particularly those using older two-cycle engines, contaminates surface waters (EPA 1998). An estimated 345 million motor boating trips, and 29 million jet skiing trips occurred in the US during 1994-95 (Cordell et al. 1997). Only a small fraction of these trips were to waters within the National Forests. Nearly all personal watercraft and outboard motors utilize "two-cycle" engines. The fuel-inefficient design of two-cycle outboard motors is essentially unchanged since the 1930s. Up to 30 percent of the gas used in the motor, goes into the water unburned. Similarly, 10 percent of the fuel used a personal watercraft, such as a Jet Ski, leaks into the water. Moreover, a personal watercraft operated for seven hours produces more smog-forming emissions than a 1998 passenger car driven for 100,000 miles (California Environmental Protection Agency 1998). The impact of marine engines on air and water quality is especially serious on summer weekend days, when smog levels are highest and use of marine engines is heaviest. To assess the impact of two-cycle motor boat engines in water quality and aquatic life, scientists measured fuel residues in waters of the Lake Tahoe Basin. They found MTBE, BTEX (benzene, toluene, ethylbenzene, xylene) and PAHs near shore in lakes that allow motorized watercraft. In open waters, the concentrations were at or under the analytical detection limit. In 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 et al. 1998).

Inefficient two-stroke carburated engines used in personal watercraft and as outboard motors are the main source of fuel pollutants. These engines emitted more than 90% of the MTBE, 70% of benzene and 80% of toluene to Lake Tahoe while they consumed only 11-12% of the total fuel. In contrast, four stroke inboard fuel injected engines emitted an estimated 8% of MTBE, 28% of benzene and 17% of toluene but consumed 87% of the fuel used by boating in Lake Tahoe. Estimated gallons of constituent load for Lake Tahoe during the 1998 boating season from two-stroke engines was in the order of thousands of gallons for MTBE, hundreds of gallons of benzene, and ten's of hundreds of gallons of toluene. Laboratory testing of newer engine technology suggested emission from old two-stroke motors can be totally eliminated by replacing them with more efficient Ficht injected engines (Allen et al. 1998, California Environmental Protection Agency 1998). Proposed legislation moves the implementation date for stricter EPA emissions controls on personal watercraft engines up five years to 2001. However, there is estimated to be 30,000-50,000 waste disposal sites (Woldt et al. 1998). Of from 2006 (California Environmental Protection Agency 1998).

In California, no violations of EPA drinking water standards for MTBE have been reported, but the potential of contamination is present (Allen et al. 1998). The California Department of Health reported that nine water sources exceeded California's interim action level for MTBE in drinking water (35 μg/L) since 1997. The concentration of PAHs observed in Lake Tahoe water was sufficient to cause negative impacts on fish growth and zooplankton survival in laboratory tests. However, there are no studies assessing PAHs emissions and their effects in the ecosystem. There was no evidence that MTBE or BTEX were transported to the bottom of the lake or accumulated there (Allen et al. 1998).

Because marinas are located right 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 storm water runoff from parking lots (U. S. Environmental Protection Agency 1993). Moreover, physical alteration of shoreline, wetlands, and aquatic habitat during the construction and operation of marinas may result in poorly flushed waterways.

When caring for boats, a significant amount of solvent, paint, oil, and other pollutants potentially can seep into the ground water or be washed directly into surface water. The chemicals and metals in paints used to protect boats, generally, have 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 tend to attach stubbornly to sediments. Hyrdocarbons persist in aquatic ecosystems and harm the bottom-dwelling organisms that are at the base of the aquatic food web. EPA recommends boaters the use of 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, keeping motors well tuned prevents fuel and lubricant leaks and improves fuel efficiency in boats (U. S. Environmental

Protection Agency 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. Fecal contamination from the improper disposal of human and pet waste may have pathogen bacteria, protozoans and viruses (Gelt 1995, U. S. Environmental Protection Agency 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, when large amounts of fish-cleaning remains are improperly disposed, the water quality deteriorates. It is important to maintain marinas with adequate wastewater hook-ups for boats and disposal sites for solid waste from boats. Well kept toilet facilities, designated pet areas, and health education postings would promote public health.

The setting and design of marinas are two of the most significant factors impacting water quality. Inlets had higher concentrations than the main channel, suggesting that hydrology plays a role in the distribution of the pollutants (Mastran et al. 1994). Poorly placed marinas disrupt natural water flushing and cause shoreline soil erosion, habitat destruction, and consequently degradation of water quality. Marinas need located and designed to be regularly flushed by natural currents. Ground cover planting and structural stabilization measures can help prevent erosion during and after marina construction. Implementing pollution prevention strategies and properly containing hull maintenance areas can control storm water runoff. Marina fueling and sewage collection stations should be maintained and designed to make cleanup of spills easier. Well designed marinas should offer an optimum combination of capacity, services, and access, combined with minimum environmental impacts and onsite development costs (U. S. Environmental Protection Agency 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 could add significantly more. Human feces may harbor virus, 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 virus are hardier and can be controlled only with additional amounts of disinfectant. Pathogenic protozoans pose the greatest risk to human health; *Cryptosporidium* and *Giardia* are small enough to pass filtration systems and up to 50,000 times more resistant to disinfectant than bacteria. In addition, the protozoan *Microsporidia* and the bacterium *Helicobacter pilori* are emerging as water contaminants that may need to be monitored in drinking water supplies (Gelt 1998).

Water that is accidentally drunk while wading or swimming poses greater 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 since 1988 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 from 17 states were attributed to recreational water exposure.

Diseases caused by *Escherichia coli O157:H7* were associated primarily with recreational lake water. *Cryptosporidium* and *Giardia* were associated with a few outbreaks in swimming pools. The outbreaks of Cryptosporidium affected almost 10,000 persons, and occurred in swimming pools that were chlorinated (Levy et al. 1998).

It is difficult to assess 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 two or three 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). The Unites States Center for Disease Control acknowledges that the very small number of cases of waterborne diseases reported probably under-represent the true picture. Epidemic outbreak of waterborne disease has been recognized only after thousands of acute cases were reported (Levy et al. 1998). Isolated and chronic waterborne diseases probably go undetected or unrecognized.

Methods used to detect enteric pathogens are not always sensitive to low concentrations; yet, very small numbers of microbes can cause illness (Gelt 1998). Routine microbiological testing may miss transient contamination by swimmers. A number of measures can be done to minimize fecal contamination: changing tables for infants in locker rooms, providing adequate toilet and hand washing facilities, posting signs against drinking water or defecating in the water, and recommending that children with gastrointestinal illness not to swim. Unfortunately, other mammals defecate in the waterbody which may introduce enteric pathogens. Hence, fecal contamination cannot be completely eliminated.

Winter Recreation

Issues and Risks - The increasing public demand for winter sporting opportunities has led to rapid expansion or creation of skiing resorts within forested watersheds (Brooke 1999). That expansion may alter the water quality of pristine environments. The National Ski Area Association estimates 60 percent of all downhill skiing in the United States occurs on National Forest lands. 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 FY 1997. The goal of the ski industry is to extend the ski season or even have the ski resorts open year around (Hoffman 1998). Some ski resorts are proposing to develop facilities for other outdoor recreation activities such as golf, swimming, and tennis. With ski resort expansion comes real state development. To maintain predictable revenues in spite of unpredictable weather, ski resorts relay 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 the natural hydrological cycles (USDA Forest Service 1991), increase traffic congestion, and are magnets for urban sprawl, all of which may impair water quality.

Findings - To satisfy the public demand and to be competitive with other providers, forest managers are authorizing the development or expansion of ski resorts. Just between January 1997, and January, 1999, the EPA Office of Federal Activities filed environmental impact statements for work in twelve ski resorts inside of National Forests. Development of ski resorts includes new construction or expansion of parking lots, service roads, downhill ski runs, cross-country ski trails, snowmobile trails, chair lifts, lodges, rest rooms, ski patrol facilities, ski school, ski repair shop, stores, hotels, and restaurants (USDA Forest Service 1990, 1991, 1992, 1997, 1998a). The construction and operation of ski facilities affects drinking water sources to different degrees. Clearing of vegetation to put ski runs on slopes (Hoffman 1998) increases the chances of soil erosion, higher turbidity and sedimentation in streams at melt-time (USDA Forest Service 1990). 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 five to ten thousand additional visitors a day. The typical average consumption rate of water at ski areas is 10 gallons per day (GPD) per skier capacity; if water conservation measures are in place, the intake could be reduced to 7 GPD. A ski resort with 13,000 skiers may need between 94,500 and 135,000 GPD (USDA Forest Service 1998a). At the same time, a small but irretrievable loss of ground water may occur due to evaporation and sublimation from snowmaking (Hoffman 1998). To prevent artificially drastic pulses in downstream flow and to maintain channel stability, ski resorts stop making snow when natural water levels are too low, or use water stored in ponds or lakes (USDA Forest Service 1997)

Ski resorts are often located in 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 groundwater flow can carry microbes and other pollutants for long distances (US Environmental Protection Agency 1999). One problem particular to ski resorts is 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 many storage ponds and apply wastewater treatment in warmer weather. Storage is not always economically or logistically feasible. One method being tested makes artificial snow from wastewater and stores it on slopes were 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 events are more likely, thus possibly contaminating surface and ground waters with effluent.

Ski resorts rely more and more on making and grooming the snow to attract skiers. Some snow making operations require pumping massive amounts of stream water. Relocating stream channels, excavating wells, or constructing ponds occur to increase the availability of water for snowmaking. The diesel generators that power the snow machines in one resort in Vermont are the eighth-largest air polluters in the state (Hoffman 1998).

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

The US EPA and CDC recognized that waterborne diseases are common in the United States (Levy et al. 1998), but data on their occurrence is very sparse. The U.S has collected incidence of waterborne diseases only since 1985. Centers for Disease Control. EPA and CDC are conducting a series of pilot studies to produce the first National Estimate of Waterborne Disease Occurrence (U. S. Environmental Protection Agency 1998). This research will quantify the magnitude of infectious disease in the United States that can be attributed to drinking water. Of particular importance are the levels of disease associated with drinking water supplies, that otherwise meet state and federal drinking water standard. This research would also serve as springboard for more localized assessments of drinking water quality.

Ability to Extrapolate Findings - A simplistic first approximation is to consider the expansion of concentrated recreation in the forests as small-scale urbanization. However, it is important to keep in mind that the toxicity of some pollutants likely produced by recreational activities has been measured only in the laboratory. Furthermore, survey data of impacts on water quality by recreation are mostly from waters that are not used for drinking (Cox 1986, Gelt 1995). The extremely varied ecology of each forest together with the diverse nature of recreation activities requires specific analysis for each situation. Even though drinking water sources appear to be not susceptible to long-term degradation as a result of recreational use, some lakes and well waters are probably susceptible to episodic local pollution (Peavy and Matney 1977). Environmental Impact Statements (EIS) are prepared when designing development of recreation resorts. Some of these EIS's present monitoring plans to evaluate surface and ground water. In the absence of specific studies, analysis of these data could be the first steep to describe regional or national patterns. Monitoring data could be used also as basis for designing future research.

Policies dealing with recreational uses of drinking water sources vary throughout the world. Decisions affecting the recreational use of a body of water are complex and have to satisfy user groups with different interests. Australia bans all recreation in drinking water reservoirs. In the United States, some utilities ban all water recreation, while others only limit full-body contact recreation (swimming and water skiing) but allow fishing, boating and wading. Moreover, there is a wide range for fecal coliform levels used as criteria to close bathing beaches. Even though limited research exists on the effects of swimming in water supplies, the American Water Works Association's official position is that no body-contact recreation should be allowed in drinking water reservoirs.

Current information is insufficient to assess the impact of contaminated water on different sections of the population visiting different forests and grasslands. It is known that Americans consume tap water containing microorganisms, trihalomethanes, arsenic, radon, lead, and pesticides. Children are at particular risk from drinking water contaminants. Children consume two and a half times more water as a percentage of their body weight than adults, and federal standards for pollutants were set based on anticipated effects on adults. The growing population of persons with compromised autoimmune systems is growing, at the same time some pathogens are becoming resistant to disinfection methods (Mott et al. 1997).

Secondary Linkages - Because recreational visits are steadily increasing, the issue of concentrated recreational impacts on drinking water supplies within the National Forests is tightly linked to urban development (USDA Forest Service 1992, USDA Forest Service 1997, USDA Forest Service 1998a). Thus the principles of urban hydrology could be applied to expanding recreational resorts. The literature on hydrology of mountainous regions and snow hydrology is relevant when studying the effects of ski resorts on water supplies; whereas, the limnological literature can be used to understand the effects of concentrated recreation in lakes, reservoirs and other surface waters. Studies on effects of recreation on forest soils, vegetation and estuaries document the impact of humans in natural ecosystems (Cole 1994). Publications on microbiology, environmental health, and public health science provide relevant information on waterborne diseases.

Ability to Address Issue **-** General principles of urban water pollution are applicable in expanding recreation resorts in all regions. However, data to quantify impacts in specific sites is 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 recreation on drinking water supplies.

Research Needs - The field of Recreation Ecology is relatively new. It is only recently that scientists are studying the relationships between use-related, environmental and managerial factors (Marion 1996). Evaluation 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 recreational resorts. One basic task is to document what kind of data have been collected as part of the routine water quality monitoring and sanitary engineering operations in forests. The next step is to design a sampling program that would specifically evaluate impacts of recreation on drinking water supplies.

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Chapter 7

Effects of Wildlife, Water Fowl, and Fish Populations and Management

It is axiomatic that the health and vigor of wildlife and fish populations is directly linked to the quantity, delivery, and quality of water. In general, activities that benefit wildlife and fish also help maintain - or at least do not significantly degrade - water quantity and quality for domestic use. Individual activities, however, have the potential to impact water supplies and uses. In this chapter, we examine some of these activities and discuss the mechanisms that may lead to changes in the quality of domestic water supplies.

Fish and Aquatic Organisms

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Introduction - Freshwater fish management that has the potential to impact water quality includes manipulations involving fish and other organisms and all activities that affect the physical, chemical, and 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 re-establish 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 (physical) elements, typically 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 cultural production facilities are among the most conspicuous potential sources of impacts to water quality related to fish management. Untreated effluent from these facilities typically consists of metabolic waste products and solids derived from uneaten fish food and fish wastes. 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 et al. 1983). 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. The results of monitoring water quality 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 et al. 1984). As a result, some investigators recommend that models be used to predict eutrophication and other environmental impacts from culture facilities (Phillips et al. 1985), particularly for cage culture operations in lakes.

Findings - The quality of effluent water varies considerably depending on the specific features of the culture system including the species of fish, intensity of production, diet and feeding regime, and the temperature and chemical character of source water (Axler et al. 1997.) The 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 60-75% of the nitrogen and phosphorus in fish feed ultimately becomes part of the waste stream (Axler et al. 1997). Unless intakes for domestic water supplies are located immediately downstream from the outfall, 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 were within thresholds for lethal exposure for aquatic organisms. Dissolved oxygen levels also decreased but were typically above 7.0 mg\L. 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 less than 0.1mL/L. 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 showed significant increases in temperature, pH, suspended solids, ammonia, organic nitrogen, total phosphorus, and BOD (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 influence of hatchery effluents tends to be localized to the immediate downstream reach (Doughty and McPhail 1995). Effects on periphyton and macroinvertebrates were not detected 400m 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 waste water through settling basins before discharge (Piper et al. 1983). 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. Although fish trematodes, cestodes, and nematodes can be transmitted to people who eat certain species of raw fish, in general, 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 et al. 1996). Residues of oxytetracycline may persist in the environment for considerable time periods; residues in sediments under net pens of intensively cultured (and treated) Atlantic salmon were present at least 10 months after treatment (Capone et al. 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 (Table 6.1) 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
5. An adverse effect on the environment is unlikely.

Table 6.1 Unapproved new animal drugs of low regulatory priority for FDA

Common name	Permitted use
Acetic acid	Parasiticide for fish
Calcium chloride	Ensure proper egg hardening, aid in maintaining osmotic balance in fish by preventing electrolyte loss.
Carbon dioxide gas	Fish anesthetic
Fuller=s earth	Reduce adhesiveness of fish eggs to improve hatchability
Garlic (whole)	Control of helminths and sea lice in marine salmonid fish
Hydrogen peroxide	Fungicide
Ice	Reduce metabolic rate of fish, especially during transport.
Magnesium sulfate (Epsom salts)	Control monogenetic trematode and external crustacean infestations
Onion (whole)	Control external crustacean parasites and sea lice

Papain	Improve hatchability of eggs and decrease incidence of disease in fish egg masses
Potassium chloride	Relieve osmoregulatory stress and prevent shock
Povidone iodine compounds	Fish egg disinfectant
Sodium bicarbonate (baking soda)	Vehicle for introducing carbon dioxide for use as an anesthetic
Sodium chloride (salt)	Relieve osmoregulatory stress and prevent shock; Parasiticide at higher concentrations.
Sodium sulfate	Improve egg hatchability
Urea and tannic acid	Denature 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; 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 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.

Ability to Extrapolate Findings - These findings are widely applicable to flowing waters.

Secondary Linkages - Impacts related to extreme hydrologic events such as floods and droughts. Depending on the design and siting of the facility, floods in particular may affect water quality if the contents of the facility are flushed downstream. Withdrawals during droughts may affect
amounts of water available for domestic uses.

Ability to Address Issues - Existing knowledge will permit managers and policy makers to make informed decisions about he impact to domestic water supplies of culture facilities.

Research Needs - None identified.

Chemical Reclamation

Issues and Risks - Pisicidal (fish-killing) compounds have been used to sample, control or eradicate fish populations since the 1930s (Bettoli and Maceina 1996). Typical applications have included eradication of exotic species such as common carp *Cyprinus carpio* from ponds, lakes, and reservoirs and removal of non-native or Aundesirable@ species from headwater streams. With the exception of sampling programs, the objective of most pisicide 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 - Only registered pisicides may be applied to waters 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. The latter two compounds, TFM and Bayluscide, are used exclusively either alone or in combination to sample or control sea lamprey, primarily in tributaries to the Great Lakes (Marking 1992; Bettoli and Maceina 1996). In the U.S. 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 for centuries been used 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 non-target aquatic organisms usually are not affected by the concentrations of rotenone used to kill fish but higher 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, it is most toxic in clear, warm, acidic waters. Residues of rotenone were detectable in the bottom sediments of experimental ponds in Wisconsin for nearly 14 days during the spring (8 C) but decreased below the limits of detection within 3-d during the summer (22 C) and fall (15 c) (Dawson et al. 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 at 250ug/L. A strong oxidizing agent such as potassium permanganate (applied at 2.0-2.5 ppm) will detoxify rotenone when applied downstream of treatment sections in flowing waters or in lakes were managers wish to limit the area of kill.

Although rotenone has a low mammalian toxicity (Marking 1988) and relatively short half-life (at 23 C less than one day in water (Gilderhus et al. 1988), public perceptions about the use of any poisonous substance in open waters has 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 nine months after treatment of Lake Davis in California.

Similar to rotenone, antimycin is relatively nontoxic to mammals (Herr et al. 1967) but highly poisonous to fish. Antimycin toxicity in the aquatic environment varies by species, and concentration over 100 ug/L may be required to kill resistant species (Bettoli and Maceina 1996). Although its effectiveness may decrease below 10 C (Tiffan and Bergersen 1996), temperature does not greatly influence the toxicity of antimycin but high turbidity (Gilderhus 1982) and alkalinity (Tiffan and Bergersen 1996) decrease its effectiveness and persistence (Lee et al. 1971; Bettoli and Maceina 1996). Turbulence leading to oxidation and foaming also may contribute to antimycin inactivation (Tiffan and Bergersen 1996).

Reliability 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 warm- and coldwater habitats under widely varying conditions. The findings are considered highly reliable.

Ability to Extrapolate Findings - These findings are widely applicable.

Secondary Linkages - Impacts related to the public perception that pisicides influence the long-term quality of water for human uses.

Ability to Address Issues - Existing knowledge will permit managers and policy makers to make decisions about the impact of pisicides on fish populations and water quality. However, controversy likely will continue over the use of pisicides in public water supplies.

Research Needs - None identified**.**

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 either intentionally or accidentally been extirpated from aquatic ecosystems. In most situations, the impact of the restoration on water quality will be almost entirely positive as 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 when during periods of low flow and high temperature they become stressed, die, and decompose. Situations involving large numbers of a single species in a relatively small area are uncommon except when anadromous fish such as Pacific salmon and clupeids (genus *Alosa*, including American shad, hickory shad, and alewifes) congregate for spawning.

Findings - Although some aspects of water quality may be temporarily degraded by the decay products from anadromous fish carcasses, the overall impact of fish carcasses can be of 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 et al. 1989; 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 et al. 1994) and the less abundant coho salmon (Bilby et al. 1996) contribute important nutrients, particularly nitrogen to otherwise nutrient-poor watersheds. Similar relationships have been observed in Atlantic slope drainages (Durbin et al.

1979; Garman 1992). Garman and Macko (1998) found that predators in a freshwater tidal stream derived a substantial proportion of their biomass carbon from marine sources during the spawning run of Alosid fishes.
Accidental introductions generally are of greater concern both ecologically and for water quality. Perhaps the most troublesome if not the only accidentally introduced aquatic pest, the zebra mussel *Dreissena polymorpha*, first arrived in North America in ship=s ballast via the St. Lawrence Seaway, sometime around 1986 (Hebert 1986). Since that time, it has invaded all of the Great Lakes and the major rivers of the eastern U.S. (Ludyanskiy et al. 1993). Invasion of virtually all major river systems in North America is viewed by some as inevitable, with only the specific timetable subject to question (Morton 1997). Although zebra mussels will attach to virtually any surface, including ships, which also act as dispersal agents (Keevin et al. 1992), the primary economic cost of the zebra mussel invasion has been the fouling of intakes for raw industrial and potable water. Zebra mussels exact a very high and escalating economic cost; over four billion dollars annually may be spent on attempts to control or mitigate its impacts (Morton 1997). The ecological impacts of the invasion have been equally profound, particularly on native unionid mussels (Haag et al. 1993). For a complete review of the issues surrounding the zebra mussel invasion, refer to: @Zebra mussels: biology, impacts, and control@ edited by Nalepa and Schloesser (1993).

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 and as natural water filters 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 (Morton 1997).

Reliability of Findings - Although this area of research and restoration is relatively recent, the findings are generally considered reliable*.*

Ability to Extrapolate Findings - Restoration ecology is one of the newest branches of ecology. As such, relatively little research has accumulated and there are few long-term studies of the effects of re-introduced species. For some species, however, particularly exotic invasive species such as the zebra mussel, the likely impact on domestic water supplies can be predicted with a high degree of precision.

Secondary Linkages - Attempts to control or eradicate species, particularly with chemicals, may impact domestic water supplies*.*

Ability to Address Issues - The current body of research will permit managers to address some issues in both the short and long term but the long term influence on water quality if repeated measures to control exotic species are required are unknown.

Research Needs - Long-term studies of the impacts of both reintroduced species and exotic species on habitat and water quality. Additional research should address the impact of other species loss or reduction on water quality following introduction of exotic or extirpated species.

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 using chemically preserved materials (i.e. creosote or pressure treated wood), which is not common practice, 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 scour of channel bottoms and sides. Turbidity may increase following 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. Although observations of naturally occurring habitat undoubtedly influenced the design of stream structures, only recently have managers attempted to mimic natural structure 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 10 cm in diameter and 1.0-1.5 m long (Harmon et al. 1986; Bisson et al. 1987; Maser and Sedell 1994; Dolloff 1994). In practice or application, managers typically consider CWD to be wood that is at least 30-cm diameter with length equal to the width of the receiving stream channel. Logging residue or slash is not be used for most habitat enhancement projects as slash tend to be unstable in all but the smallest stream channels. CWD enters stream channels naturally by a number of routes including bank undercutting, windthrow, and as a result of catastrophic events (e.g. snow and debris avalanches, hurricanes). Managers add CWD by direct felling or toppling of streamside trees or by transport from more distant sources.

Findings - In general, the relatively small amounts of wood added by managers to enhance or restore habitat are not likely to exert a major impact on water quality except 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, such as in areas near log storage ponds or transfer sites, dissolved oxygen may be depleted and hydrogen sulfide and ammonia produced (Sedell et al. 1991). Leachates from logs may contain toxicants but these are unlikely to reach significant concentrations under natural conditions (Schaumburg 1973; Thut and Schmeige 1991).

Reliability of Findings - Despite being a topic of intensive research interest for only about the last 20 years, a large body of information has accumulated on the ecology and management of instream CWD. A broad range of studies, including historical observational and experimental (eg. wood removal and addition) research has been conducted in watersheds located all across North America and several other continents. Reliability is high.

Ability to Extrapolate Findings - Findings from studies conducted 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.

Secondary Linkages - Catastrophic additions of woody materials, use of non-woody materials for habitat improvements (eg. concrete or other construction rubble), use of live vegetative materials for streambank stabilization, forest best management practices, including stream crossings and streamside management zones.

Ability to Address Issues - The information available should allow managers to address most issues related to the effects of woody materials installed for fish habitat on water quality for domestic uses.

Research Needs - 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 result in a net export of nitrogen into the water body, resulting in either excessive algal blooms or the need to remove the nitrogen prior to domestic use.

Liming of Acidified Waters

Issues and Risks - Waters that drain areas where the bedrock does not contain a high buffering capacity can be acidified by major soil disturbing activities such as mining (Nelson et al. 1991) or acid precipitation. Problems associated with acidification of surface waters include low pH, which disrupts physiological processes for many aquatic organisms, and increases in toxic forms of metals such as aluminum. Left uncorrected, continuing acidification can extirpate entire faunas (Olem 1991). The quality of surface waters can be dramatically improved by treating streams, lakes, or whole catchments with a soluble basic mineral such as crushed limestone powder, sand, or gravel.

Findings - Although liming to reverse the effects of acidification is at best temporary solution and must be repeated to maintain water quality, the beneficial effects (if not the mechanism of action) of liming have been known for hundreds of years (Henrikson and Brodin 1995; Porcella et al. 1995). The primary benefit of liming has been to preserve or recover fish stocks and diversity of aquatic species. Water quality after liming generally improves as indicators such as pH rise and concentrations of metals including toxic forms of aluminum decrease (Wilander et al. 1995). Concentrations of many other trace metals also are influenced by acidification and generally increase as pH decreases (Veselý 1992). Liming may also result in the precipitation of metals and stress to aquatic species present in mixing zones, such as in downstream reaches where acidic tributaries join with limed waters (Henrikson and Brodin 1995).

Reliability of Findings - In the last 20-30 years, a large body of research has accumulated, particularly in Europe and North America, describing the mechanism, consequences, and treatment of acidified surface waters. The findings are considered highly reliable.

Ability to Extrapolate Findings -Many studies related to water quality impacts of liming have been replicated numerous times with consistent results. The findings are widely applicable.

Secondary Linkages - Forest management practices, including forest fertilization and pesticide application programs and harvest strategies and rotations. Habitat and species/population restoration projects often occur concomitantly with liming.

Ability to Address Issues - At the current scale of application (eg. individual small watersheds), current knowledge should permit managers to address most issues associated with liming

Research Needs - In general, knowledge of virtually all of the long-term (i.e. 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 for liming; from the standpoint of understanding the more immediate influence on water quality, some of the areas needing particular attention include the effects of liming on the uptake and transport of mercury and other metals in aquatic organisms, the consequences for water quality of reacidification if liming is stopped, and the social effects and consequences for domestic water supplies of liming.

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Wildlife Management

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The following description is provided to give a general background to the two pathogens that are of great interest those interested in drinking water quality.

Description of Protozoan Pathogens *Giardia* and *Cryptosporidium.*

Giardia spp. cysts and *Cryptosporidium* spp. oocysts are two parasitic protozoans, which are active and reproduce within their hosts and encyst to survive in the environment during transmission between hosts. These organisms have been recognized as significant sources of gastrointestinal illness (Jokipii, et al. 1983, Kenney 1994; Moore, Herwaldt et al. 1994). The risk posed by these parasites is believed to be significant. It has been reported that as little as one cyst can cause infection (Rose 1991,Medema et al. 1995, Rose et al. 1995) These organisms have also been found to be resistant to disinfection (Campbell, Tzipori et al. 1982; Clark and Regli 1993; Craun 1991; Haas and Heller 1990; Hoff and Rubin 1987; Jarroll, Bingham et al. 1981; Kong, Swango et al. 1988; Quinn and Betts 1993; Rice 1981; Rubin, Evers et al. 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 United States 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 1989) 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 since disinfection by chlorination is not fully effective in eliminating the threat posed by *Giardia* and *Cryptosporidium* (Clark, et al. 1989; Campbell, et al. 1982). 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 the 1996 amendments to the Safe Drinking Water Act which adds a section specifically funding additional watershed research for these organisms.

Giardia - To survive in the environment *Giardia* encysts itself into a resistant form. *Giardia* spp. cysts are 5 to 15 •m in size and oblong in shape (Figure 7.1). Early research on this parasitic protozoan identified the *Giardia* species based on the median body morphology and the host it was found in. Accordingly, *Giardia* species are G*. muris,* G. *agilis,* G*. duodenalis* and are usually found within 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 *Giardia* to be able to cross-infect different species of hosts (Meyer, 1980).

Identification of *Giardia* as a waterborne parasite for humans was first reported in the 1940•s during an epidemiological investigation of a disease outbreak in a apartment building located in Tokyo, Japan (Davis, 1948). *Giardia* has more recently been reported as the most frequently identified parasite responsible for waterborne outbreaks in water supplies in North America using surface waters (Craun, 1991). 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, 1980) and is illustrated by Figure 7.2. *Giardia* monoxenous, which means that all of its life stages of 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 was first identified by (Tyzzer, 1907) over 90 years ago as a parasite found in *Mus musculus* (common mouse). The significance of these parasites began to be recognized in the 1970's when a number of reports were published identifying *Cryptosporidium* infection as some of the cause of diarrhoeic calves (O'Donoghue, 1995). Human infections of Cryptospordiosis 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 1991; Ungar 1990).

Cryptosporidium spp. oocysts are spherical in shape and 4 to 6 •m (16 to 24 millionth of an inch) in diameter (Barer and Wright 1990; Casemore, Sands et al. 1985; Casemore 1991; Current, Upton et al. 1986; Current 1987; Fayer and Ungar 1986; Issac-Renton, Fogel et al. 1987; O'donoghue 1995; Smith and Rose 1990; Ungar 1994; Upton and Current 1985). Most oocysts contain up to four sporozoites (free living form). A number of species have been identified among various hosts. Many of these species have been found to be able to cross-infect different species of hosts. Several species of the *Cryptosporidium* species are found more often in association with specific host species especially when the host species are within the vertebrate class (i.e., *C. muris* in mammals, *C. meleagridis* in birds, *C crotalia* in reptiles and *C. nasorum* in fish; (Levine 1984; O'Donoghue 1995).

The life cycle for *Cryptosporidium* has been described by (O'Donoghue, 1995) and (Current and Bick 1989) and is illustrated in Figure 7.3. Although its life cycle is similar to Giardia in that *Cryptosporidium* encysts to survive outside its host and that it is also monoxenous (all life stages occur within the infected animal), the cycle differs by being more complex due to the addition of a sexual stage of reproduction within the host. The oocyst of *Cryptosporidium* 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; Robertson, et al. 1993). 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 (form of *Cryptosporidium* that reproduces asexually) 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 fertilizes 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 to complete the life cycle.

Introduction - Although a number of reports in the literature indicate possible drinking water quality impacts from wildlife, there is no readily available studies to either qualify nor quantify these impacts. By far, the majority of the evaluations has been conducted with waterbirds and is presented in the next section. The studies that are available tend to base comments on wildlife impacts to water quality on intuition since almost no information is available to quantifying this potential source of pollution.

There may be unpublished studies and agency reports that are not readily available that provide greater evidence. Some of these are discussed in Chapters 4 under Range Management and 6 Recreation. Some inferences can be drawn for wild ungulates (Hoofed animals) from the discussions presented in Chapter 4 on grazing by livestock. However the reliability of the application of this information to wildlife is questionable due the lack of supportive studies.

Erosion, turbidity, nutrients and sedimentation

Issues and Risks – As with the issues identified in Chapter 5 on rangeland management, overuse of available resources in an ecosystem by any species can result in significant erosion. Increases in turbidity, nutrients and sedimentation tend to be associated with greater erosion. Inadequate predation to control herbivore populations can result in significant losses of groundcover vegetation. Such losses tend to allow for increase surface runoff volumes and velocities. These increases in turn accelerate erosion of the land surface and create higher peak flows in streams and rivers causing bank erosion.

Findings from Studies - A review of the literature provided only one study that specifically studied the impacts of wildlife on nutrients and turbidity levels in water (E. Gereta and E. Wolanski, 1998). Unfortunately, this study may have limited applicability due to its location in Africa. However, it does supply some applicable data on generally observed water quality impacts from wildlife. The study included measurements of the levels of pH, eutrophication, dissolved oxygen, and light penetration during the wet and dry seasons of 1996 in the southern plains of the Serengeti National Park in Tanzania, Africa. It attributes the heavily eutrophicated water bodies observed during the dry season to the input of animal dung. The euphotic zone was noted to be less than 1 cm thick and the waters generally inhospitable to aquatic life. Also noted in this study are the positive effects such wildlife as hippos, crocodiles and various mammals in preventing the formation anoxic conditions at river bottoms by stirring and mechanical aeration.

Human pathogens

Issues and Risks - Wildlife can act as a reservoir for human pathogens. The historic focus for the pathogen issue with regard to drinking water quality has been with total and fecal coliform bacteria with a specific emphasis on *Escherichia coli* (E. *coli*) as general indicators of fecal contamination. A new emphasis within the last decade has been placed on the impact from the parasitic protozoans *Giardia* and *Cryptosporidium*.

Findings from Studies - Most of the information on pathogen and wildlife does not involve water quality impacts. Studies on this topic generally provide reports on the occurrence of specific pathogens in specific wildlife species. Table 7.2 presents some of the animals that have been reported to be hosts to *Giardia* and *Cryptosporidium* (Wade 1999). A focus has been placed on *Giardia* and *Cryptosporidium* since these pathogens have been identified by EPA as the species most threatening to water supplies.

These human pathogens can be transmitted by a wide range of species and is not just exclusive to mammals.

Several reports appear in the literature that attributes waterborne outbreaks of human disease to wildlife. (Geldreich, 96, Georgi, 1986, Frost 1980, Davies, 1979, Stuart et al 1971) These findings are generally based on conjecture drawn from observing elevated pathogen levels then using a process of elimination to determine the possible sources of the pathogens. None of the studies include data that quantifies observed pathogen infection in wildlife. An older record exists for *Giardia* due to its earlier identification as a protozoan that causes human disease. Several studies have indicated beaver and muskrats as reservoirs for *Giardia* cysts. (Craun 84, Jackubowski, 84, Frost, 1980, Kirner,1978, Lippy, 1978). More recent studies have been focused on Cryptosporidium. (Add more discussion - coordinate with recreation section- note refs by Suk87 and Stuart 71)

Reliability of Findings - Only rudimentary studies have been conducted with regard to wildlife and water quality impacts. Almost all the research with regard to these two subjects has revolved around the impacts that water pollution has had on wildlife health and reproduction. Accordingly, the reliability of the findings is poor since the reported studies have not quantified impacts.

Ability to Extrapolate Findings - Since little information is known about wildlife impacts on water quality, it may be useful to generalize the limited findings from regionally specific data. Additionally, some generalizations may be made with information from the impacts and management from domesticated animals. However, these generalizations should be evaluated with caution until more information becomes available.

Secondary Linkages- N/A

Ability to Address Issues - It's clear from the literature that there is insufficient information to make complex decisions on how wildlife impacts water quality. There is some anecdotal evidence that wildlife may be a significant source of contaminants to drinking water, but this information is too limited allow sound decision making. Intuitively, wildlife has the potential to contribute to drinking water pollution. However, further research is needed to determine the significance of this threat and the effectiveness of management practices.

Research Needs **-** Additional scientific study is needed to determine how significant wildlife contributes pathogens and other pollutants to water supply watersheds. Only recently has there been an interest to begin to explore this topic with regard to compliance with drinking water standards. An outline of the topics that should be given a high priority for future investigation is provided in the following sections.

<u>Complete identification of wildlife species that are reservoirs for waterborne human pathogens</u> – Reported cases of wildlife infected by human pathogens has been incidental. Research is needed for a systematic inventory of the species of wildlife that carry waterborne human pathogens. Along with this identification, is the need for information on the range of population densities for the important carrier species. Such a population estimate needs to be specific for the habitats these species are found in.

<u>Study the significance of contaminant loads from wildlife</u> – Although anecdotal information is available to indicate that wildlife may have a significant impact on water quality, there have not been any studies to attempt to quantify this impact. This will likely require new sampling and statistical techniques.

Included in determining this significance, should be laboratory studies that determine concentrations of nutrients and pathogens in the feces and daily volume output for each species of concern.

Study how wildlife population and population dynamics change with habitat changes – With improved understanding of the significance of the threat posed by wildlife, research can be undertaken to improve our understanding on the changes to species populations with changes to habitat.

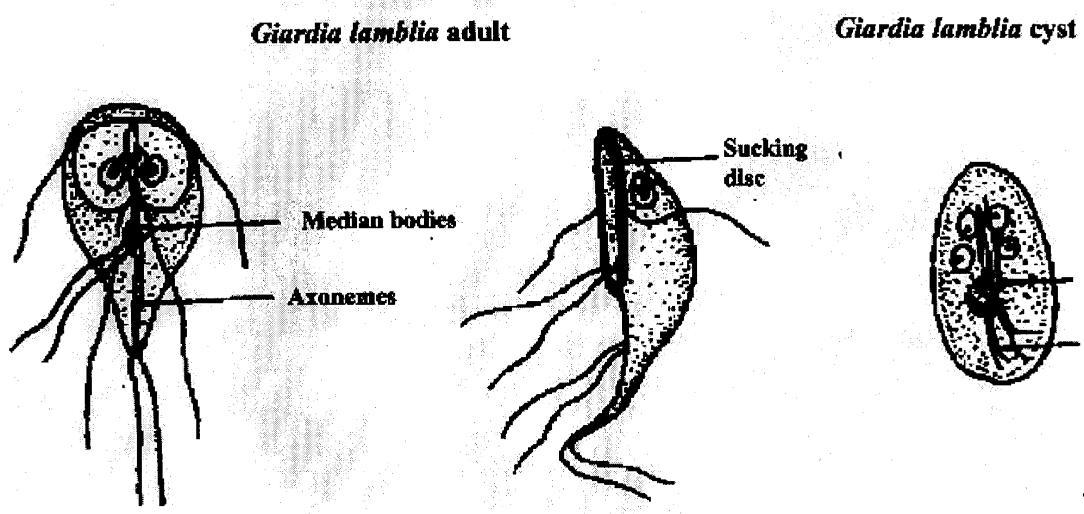

Figure 7.1. Giardia.

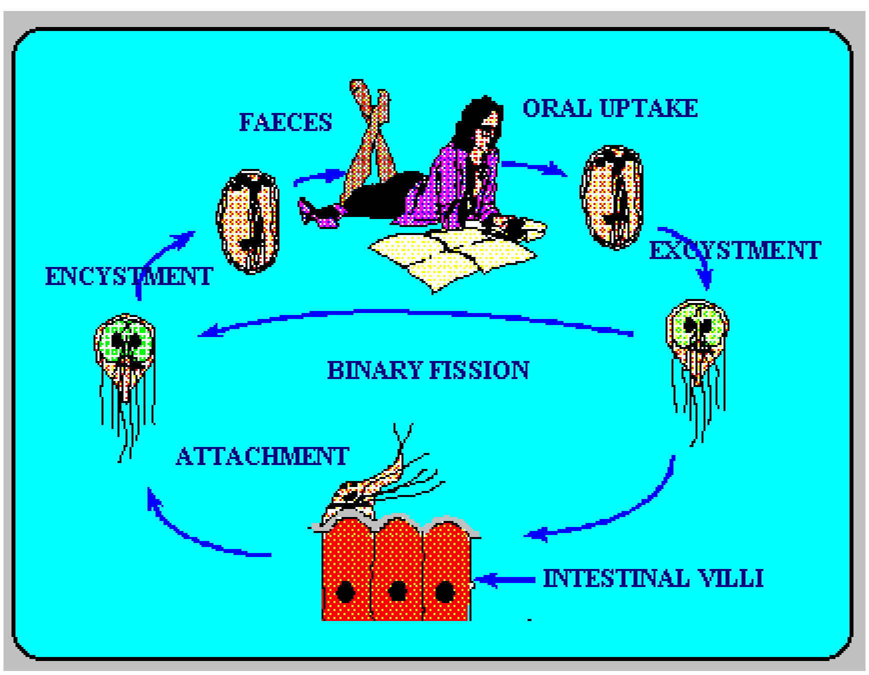

Figure 7.2. Giardia life cycle

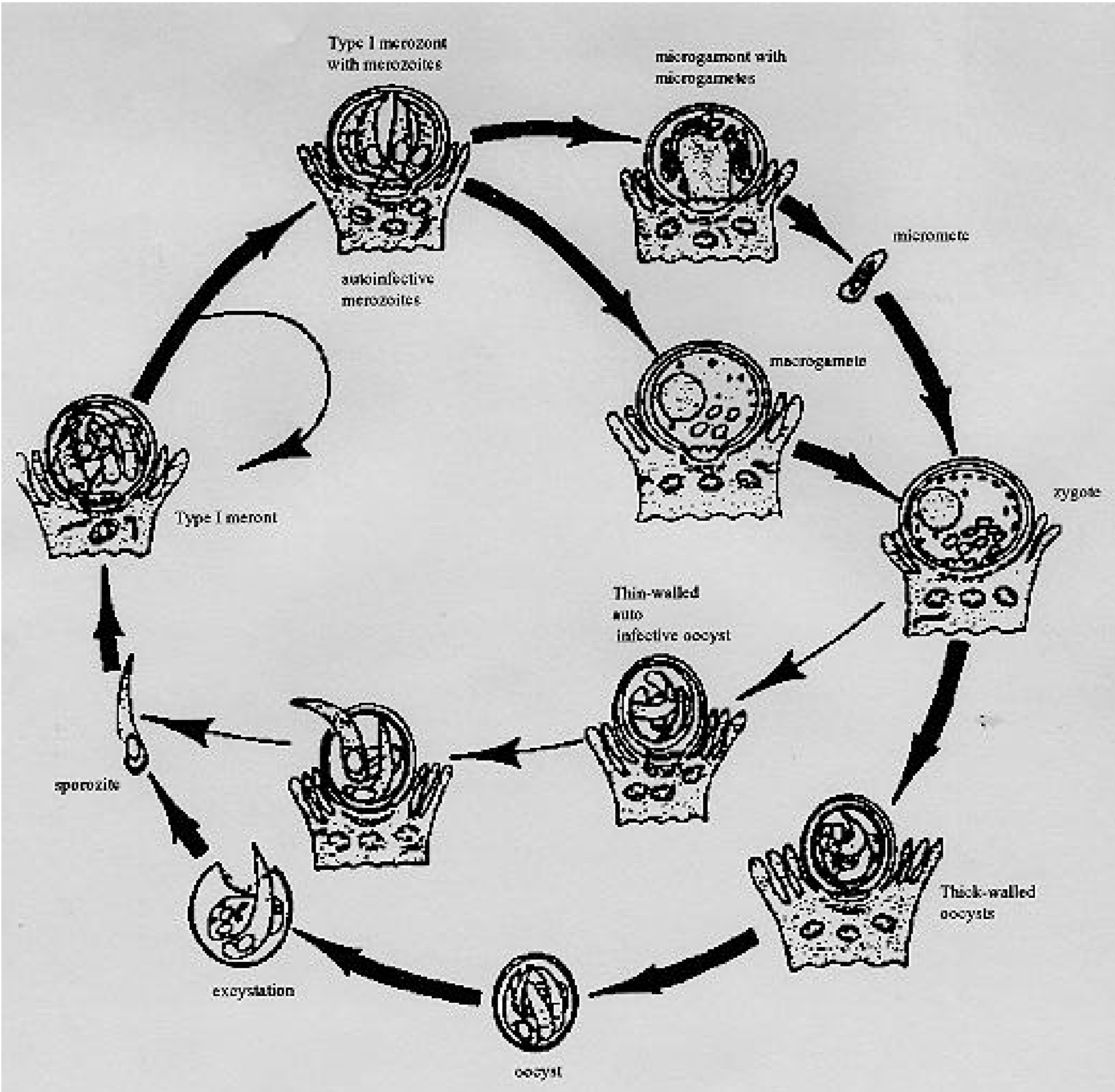

Figure 7.3. Crypto life cycle including sporozoite and oocyst pictures

Table 7.2. Species reported as hosts to protozoans *Giardia* and *Cryptosporidium* (adapted from Wade, 1999)

Species	Common Name	Parasites Hosted
Pices		
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*
Bufo regularis	Common Toad	*Giardia*
Rana pipiens	Leopard Frog	*Giardia*
Rana clamitans	Green Frog	*Giardia*
Reptilia		
Chelonia mydas	Green Turtle	*Cryptosporidium*
Geochelone elegans	Star Tortoise	*Cryptosporidium*
Geochelone carbonaria	Red-footed Tortoise	*Cryptosporidium*
Squamata Lacertilia (lizards)		
Agama aculeata	Kalahari Spiny Agama	*Cryptosporidium*
Agama 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*
Elaphe o. obsoleta	Black Rat Snake	*Cryptosporidium*
Elaphe o. quadrivittata	Yellow Rat Snake	*Cryptosporidium*
Elaphe o. lindheimeri	Texas Rat Snake	*Cryptosporidium*
Elaphe guttata	Corn Snake	*Cryptosporidium*
Elaphe v. vulpina	Western Fox Snake	*Cryptosporidium*
Gonysoma oxycephala	Red-tailed Green Rat Snake	*Cryptosporidium*
Pituophis melanoleucus	Black Pine Snake	*Cryptosporidium*
Pituophis melanoleucus catenifer	Gopher Snake	*Cryptosporidium*
Drymarchon corais couperi	Eastern Indigo Snake	*Cryptosporidium*
Lampropeltis zonata pulchura	San Diego Mountain Snake	*Cryptosporidium*
Lampropeltis triangulum	various subspecies	*Cryptosporidium*
Nerodia h. harteri	Brazos Water Snake	*Cryptosporidium*
Nerodia r. rhombifera	Diamondback Water Snake	*Cryptosporidium*
Boiga dendrophila	Mangrove Snake	*Cryptosporidium*
Crotalus horridus	Timber Rattlesnake	*Cryptosporidium*
Crotalus atrioca udatus	Canebrake Rattlesnake	*Cryptosporidium*
Crotalus l. lepidus	Rock Rattlesnake	*Cryptosporidium*
Aves		
Anseriformes		
Branta canadensis	Canada Goose	*Cryptosporidium/Giardia*
Anser anser	Domestic Goose	*Cryptosporidium*
Cygnus sp.	Tundra Swan	*Cryptosporidium*
Cygnus 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*
Larus argentatus	Herring Gull	*Cryptosporidium*
Larus 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*
Ardea cinerea	Gray Heron	*Giardia*
Ardea cocoi	Cocoi Heron	*Giardia*
Egretta alba	Great Egret	*Giardia*
Egretta caerulea	Little Blue Heron	*Giardia*
Nycticorax nyctocorax	Black-crowned Night-Heron	*Giardia*
Nycticorax 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*
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 sp.	Shrew	*Giardia*
Blarina brevicauda	Short-tailed Shrew	*Cryptosporidium/Giardia*
Sorex 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*
Microtus chrotorrhinus	Rock Vole	*Giardia*
Microtus pennsylvanicus	Meadow Vole	*Cryptosporidium/Giardia*
Microtus pinetorum	Pine Vole	*Giardia*
Microtus longicaudus	Long-tailed Vole	*Giardia*
Microtus ochrogaster	Prairie Vole	*Giardia*
Microtus californicus	Meadow Vole	*Giardia*
Microtus richardsoni	Water Vole	*Giardia*
Clethrionomys glareolus	Bank Vole	*Cryptosporidium*
Clethrionomys glareolus skomerensis	Skomer Bank Vole	*Cryptosporidium*
Apodemus sylvaticus	Wood Mouse	*Cryptosporidium*
Rattus rattus	Roof Rat	*Cryptosporidium*
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*
Peromyscus maniculatus	Deer Mouse	*Cryptosporidium/Giardia*
Clethrionomys gapperi	Red-backed Vole	*Cryptosporidium/Giardia*
Pitymys savii	Savi's Woodland Vole	*Giardia*
Rattus rattus	Ship Rat	*Giardia*
Rattus 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*
Spermophilus beecheyi	Ground Squirrel	*Giardia*
Spermophilus richardsoni	Richardson's Ground Squirrel	*Giardia*
Spermophilus 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*
Mustela 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*
Mustela 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*

Waterbird Populations

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Introduction - This section synthesizes the available information on the sources, mechanisms and risks of drinking water contamination from waterbird to be used as a guideline for management of potable water supplies located within forested lands. Waterbird, which include ducks and other duck-like swimming birds, gulls, and wading birds have been implicated in the contamination of both large and small drinking water supplies, particularly since the implementation of the Clean Water Act, Surface Water Treatment Rule (SWTR) of 1989 (EPA 1999) and the Interim Enhanced Surface Water Treatment Rule (IESWTR) of 1998 (EPA 1999) set forth by the Environmental Protection Agency (EPA). SWTR required all municipal water suppliers to comply with tougher water quality parameters such as the amount of bacteria (fecal coliform bacteria) that is allowed in pre-treated surface water at all municipal water suppliers. As a result, municipal water supplies developed and implemented watershed protection programs to identify and mitigate all sources of biological pollution.

Risks and Issues **-** The United States Department of Agriculture (USDA), Wildlife Services collects and disseminates information concerning nuisance-related problems with a variety of birds. Waterfowl (geese, swans and ducks) and gulls commonly create human health concerns by polluting potable drinking water supplies and fowling recreational areas such as swimming beaches (USDA undated). Species like the cormorant are thought to negatively impact recreational fisheries and drinking water supplies (reference). A variety of pollutants can be expected from waterbird in relation to their behaviors or activities on or near a human drinking water supply. These include biological, chemical and nutrient pollutants that may be released or deposited by waterbird. The type of pollutant will determine what kind and level of waterbird management required to prevent contamination to a reservoir system.

Estimating the degree of impact from the various types of contamination requires information on the species of waterbird inhabiting the water supply, number of birds per species, daily defecation or pellet regurgitation rates, amount of contamination per specified weight of feces or pellet, and the daily amount of activity by each bird on the water (Table 7.3).

Findings from Studies – Studies have found that water foul affect the biological, nutrient and chemical quality of water.

Biological

Among the various types of waterbird pollutants to a drinking water supply, the biological types from waterbird excrement containing bacteria, protozoans or enteric viruses, are routinely monitored by water suppliers and regulated by the EPA and various State Health Departments. Numerous studies have documented the occurrence of fecal coliform bacteria and other pathogens in many North American waterbird species (Ashendorf et al 1997, Gould & Fletcher 1978, Conover & Chasko 1985, Wadleigh 1968, Beddard & Gauthier 1986, Benton et al 1983, Damare et al 1979, Hussong et al 1979, Standridge et al 1979, Barnes 1972, Geldreich et al 1962, Berg and Anderson 1972, Faddoul 1966, Fennell et al 1974, Williams et al 1976, Branvoldm et al 1976, Neilson 1960, Jones et al 1978, Summers 1967). As a result, bird control

Table 7.3. Formula for Estimating Contaminant Loads to a Reservoir System

Estimate of biological contaminant or nutrient load to a reservoir from water birds is a function of:

Species richness and eveness
(Number of species and number
of individual per species)

+

Species activity time on or over water
(Timbe spent by each individual/species
on or over the reservoir surface suspected
of defecating or releasing pellets)

+

Species daily defication/release rate
(Daily fecal output or pellet mass released
per species)

+

Concentration of excrement/pellet released
(Amount of contaminant per mass of feces or pellet)

programs have been developed and implemented at many municipal water sources (Amling 1980, Blokpoel & Tessier 1984, DEP 1994 - 1999, MDC 1994). Among the known human pathogens monitored, waterbird excrement has been reported to contain both human and non-human pathogens (DEP ?). 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 (Table 7.3). The origin of fecal coliform bacteria is solely from warm-blooded animals (Clesceri 1994) and analytical methods to identify the origin of bacterial type (human versus non-human) 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, methods are being developed to link bacterial sources from water samples to either human, non-human or both sources (DEP ?).

Nutrients

Nutrient contributions of nutrients to aquatic ecosystems by waterbird has been well documented (Scherer et al 1995, Bazely and Jefferies 1985, Gould and Fletcher 1978, Manny et al 1994, Noyes 1981, Gwiadza 1996, Merola 1987, Merola 1988, Norwell and Frink 1975, Manny et al 1975, Gere and Andrikovics 1994, Gere and Andrikovics 1992), however, debate continues over significance of impacts from bird defecation on nutrient loading (Murphy et al 1984, Hoyer

Table 7.3. Indicator and Principal Pathogens of Concern in Contaminated Drinking Water (Murray et al 1995 and Hurst et al 1997)

Organism type	Disease/Symptoms
Fecal Coliform Bacteria Group	
Bacteria	
Escherichia coli	Diarrhea, Dysentery, Hemorrhagic Colitis/Diarrhea Nausea, cramps, fever, vomiting, mucus in stool
Klebsiella pneumonae	
Important Human Pathogenic Organism Suspected from Feces	
Bacteria	
Shigella spp.	Shigellosis/Diarrhea, fever, cramps, tenesmus, blood in stools
Salmonella typhimurium	Salmonellosis/Abdominal pain, darrhea, nausea, vomiting, fever
Salmonella typhi	Typhoid fever/Abdominal pain, fever, chills, diarrhea or Constipation, intestinal hemorrhage
Enterotoxigenic	Diarrhea/Diarrhea, fever, vomiting
Campylabacter Jejuni	Gastroenteritis/Abdominal pain suggesting acuter appendicitis, fever, headache, malaise, diarrhea, vomiting
Vibrio cholerae (Incidence is rare or negligible in U. S.)	Castroenteritis/Vomiting, diarrhea, dehydration
Yersinia sp.	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 27nm 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 indivuduals may develop chronic diarrhea
Entamoeba histolytica	Amebiasis/Vary from mild diarrhea with blood and mucus to acute or fulminating dysentery with fever and chills

and Canfield 1994). Among the various nutrients identified in waterbird excrement, Nitrogen and Phosphorus are of greatest interest. Phosphorus, a limiting nutrient for plant growth, and Nitrogen have the potential to increase the eutrophication process and degrade water quality at a drinking water supply (Vollenweider & Kerekes 1980). Eutrophication, from nutrient enrichment, often results in excess growth of algae and other aquatic vegetation. Algal growth potentially increases the release of dissolved organic matter, which when combined with chlorine for water purification may form a halogenated hydrocarbon compound (DEP 1993). It is during this chlorination treatment process that, combined with Phosphorus, can produce harmful side effects. EPA cooperatively works with individual States to develop and regulate municipal water suppliers for total maximum daily loads or TMDL's, which is defined as an estimate of a pollutant, such as P, entering a reservoir from all potential sources.

Chemical

Another type of waterbird contamination may be of chemical origin (i.e. glandular releases, body oils, pesticides, hydrocarbons transported by feathers, etc). The impact to drinking water supplies from glandular releases and body oils emitted by waterbird are not well studied. Chemical pollutants carried externally on feathers or vectored by waterbird, 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, therefore, gulls will seek alternative locations for foraging such as agricultural areas, urban centers, moist fields, landfills and others where birds may be exposed to a variety of chemicals either through ingestion or external attachment (reference). At these foraging locations, gulls may accumulate various chemicals on their feathers and transport them back to the reservoir to nightly roosting activity. Gulls may also ingest contaminated materials from landfills or sewage treatment facilities, carry them back to the reservoir and regurgitate pellets with these chemicals. Additional studies are needed to determine these impacts.

Water Foul as Vectors of Contamination

A variety of human-related activities, such as urbanization, resource exploitation, agriculture and land conservation/preservation principles have the potential to promote or discourage populations for variety of waterbird species.

Waterbird sources of pollution to a drinking water supply system generally include direct deposition from roosting, foraging and flyover activity. Activities outside the reservoir or watershed might include foraging at agricultural operations where birds may acquire pathogens, such as *Giardia* spp. or *Cryptosporidium* spp. from domestic farm animals (Graczyk et al 1998) and foraging at urban centers such as shopping malls or landfills where numerous sources of chemicals can be amassed. Agricultural operations such as the spreading of fodder or manure, growing crops, tilling soil exposing soil organisms offer attractive foraging locations for some species that travel great distances like gulls (i.e. as much as 30 to 40km/day for ring-billed gulls (Bull 1979)). In addition to fecal coliform pollution, protozoans such as *Giardia* spp. and *Cryptosporidium* spp. are often regulated by water suppliers and have been associated with excrement releases by Canada geese, gulls and other wildlife (Graczyk et al 1998, O'Donoghue 1987). In a study of Canada geese foraging on agricultural lands near the Chesapeake Bay, researchers identified a high incidence of *Giardia* and *Cryptosporidium* spp. in their fecal matter (Graczyk et al 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 (Cornell University/DEP in preparation).

The pathways by which water contaminated with biological, chemical and nutrient can enter into the reservoir from waterbird include stream flow through drainage basins, storm-water surface sheet flow, flyover fecal releases and direct fecal deposition. The location of contamination 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 from will also affect the extend of impacts. The location factor also depends on the die-off rate, settling rate and water travel time of the type of contaminant in question. As a result, contaminant sources entering distant from a water intake chamber are less likely to impact water samples than those in close proximity.

Seasonality of Impacts

Seasonality of impacts may also play an important role in determining pollutant loadings from waterbird. Studies have demonstrated positive correlations with seasonal migratory movements (stopovers and overwintering) of species like Canada geese, ducks and gulls and increases in fecal coliform bacteria that exceed the SWTRA regulations (DEP 1994 -1999, MDC ? and others). Local breeding populations of waterbird may also negatively impact water quality. 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 waterbird populations.

Risks to drinking water supplies are often associated with seasonal increases of waterbird populations from migratory or overwintering activity. These population fluctuations may be short term migratory stopovers, or longer seasonal fluctuations due to breeding activity or overwintering. Some species, like Canada geese (*Branta canadensis*), depending on the geographic location, maintain year around residence. In the northeastern United States, Canada geese were propagated and released for sports-hunting and populating large estates during the early to mid-twentieth century (reference). As a result, local and regional goose populations expanded greatly as the migratory behavior was essentially eliminated. Waterbird species populations, such as gulls, were also artificially increased in numbers with the advent of point-source feeding locations such as landfills, agricultural properties and urban and suburban shopping centers. Other waterbird species known to negatively impact water quality include cormorants, swans and ducks (reference).

Reliability of Findings – The types of water quality impacts are understood. The magnitude of ijmpacts are less well understood.

Ability to Extrapolate Findings - Resident and migratory water foul will pose similar problems to water supply reservoirs. The magnitude of the impact will vary by species and number of birds habitating the reservoir, the surrounding environs and associated sources of contamination, and seasonal patterns of migration. Therefore, water foul impacts will vary and need to be assessed for each reservoir.

Ability to Address Issues - Natiional forest land managers may have different objectives concerning habitat management for waterbird species than a reservoir manager. Both forest managers and reservoir managers can and do impact the population dynamics of waterbird species. These management objectives often oppose one another to obtain the intended goal. For example, the forest manager may develop a forest management plan that seeks to enhance protection and breeding or migrational opportunities for certain waterbird species while the reservoir manager might try to discourage locally breeding and migrant waterbird populations creating an unattractive habitat.

The ability to address the question of risk to drinking water supplies from waterbird, located within national forested lands, requires the development of a comprehensive Watershed Protection Plan. This Plan needs to identify all sources of pollution to expedite the development of management plans to eliminate these negative impacts. If it is determined that waterbird populations do contribute pollutants to the watershed and potentially effect water quality an additional plan for waterbird management should be developed. This Waterbird Management Plan should also incorporate research and management objectives from both the land manager and the reservoir manager regarding waterbird populations. Managers should initiate population inventories and an assessment of all waterbird species that breed or migrate throughout the watershed to identify all potential impacts on water quality. The type of pollution will help focus on the potential origins of pollution including fate and transport mechanisms from source to water distribution structures. Once identified, waterbird population data can be used to concentrate on management practices to reduce or eliminate these sources of pollution without compromising local or regional species conservation plans that might impact population dynamics.

Research Needs -

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Chapter 8

Urban/Wildland Intermix Influence on Dringing Water Quality

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Introduction - This chapter specifically examines drinking water issues related to administrative facilities and in-holdings. Forest Service facilities are grouped with residential and commercial development because, structurally, facilities are similar to other developed sites. Forest Service facilities, however, are required by law to meet all current federal and state mandates, and specific standards.

Land parcels contained within public lands that are not owned by the managing agency are called in-holdings. In-holdings are owned by other federal, state and local government agencies, and by private landowners. Of particular interest is privately owned land, because of it's propensity for development. Unfortunately, there is no state or national data on the acreage of in-holdings within public lands. However, this data is available for National Forests and varies by Forest Service Region (Table 8.1). Region 4 has the least area of in-holdings with 6.9 percent. By comparison, Region 8 and 9 had 48.6 and 45.6 percent in in-holdings, respectively.

Of particular interest is how rapidly are these in-holdings being developed, but no data are available. 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 the periods 1980-1990 and 1990 to 1996 using census data (U.S. Census Bureau 1997). Between 1980 and 1990, these counties grew by 18.5 percent, while nation's population grew by 9.8 percent (Table 8.1). For 1990 to 1996, the nation grew by 6.4 percent, and the counties grew by 10.1 percent. Using 1996 census data, these counties contain 22.4 percent of the nation's population.

Population change in these counties varied by region and time period. Between 1980 and 1990, Region 5 experienced the greatest percent increase with 28.9 percent, while Region 1 decreased by 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 effect of this development on drinking water quality depends on the location within the watershed, the concentration of development and existing conditions. Unfortunately, data are unavailable to examine those variables.

Of course, this growth did not all occurred within and adjacent to public land. Nevertheless, the growth rates indicate indirectly that in-holdings are being developed for residential and recreational uses, and for commercial purposes.

Issues - During the past 20 years private lands within and adjacent to public forest and grasslands have developed rapidly for residential, commercial and recreational use. This development poses a significant threat to drinking water quality through surface and groundwater contamination. For surface waters, development occurs near the headwaters of streams where water quality is generally at it's highest and is easily degraded because of stream size. For groundwater, about 95 percent of rural communities use groundwater as the principal source of drinking water. Sources of pollution result from wastewater treatment, non-point source pollution, underground storage tanks, solid waste storage, and hazardous material storage.

In 1995, U.S. Environmental Protection Agency (EPA) summarized water quality information submitted by States and Tribes, and other jurisdictions to identified sources of pollution of surface and groundwater (EPA 1998c). For rivers, streams, lakes, ponds, and reservoirs, municipal point sources and runoff from residential and commercial sources were identified as significant pollution contributors. For groundwater, principal sources included leachate from leaking underground storage tanks, septic tanks, and landfills. Of specific importance is the effect of in-holding urbanization on the water quality of surface and groundwater sources (Table 8.2) from wastewater treatment, urban/storm runoff, underground storage tanks, and landfills.

The report identified urbanization as a major factor in polluting, 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 decrease groundwater recharge (Weiss 1995). Increased peaks flows can negatively affect drinking water quality by causing bank destabilization and streambed scouring, which increase turbidity and sedimentation (Phillips and Lewis 1995). Reduced groundwater recharge decreases base flow in streams and increase pollutant concentrations. A decreased base flow impairs aquatic habitat and riparian wetlands and increases the stream's sensitivity to other pollution inputs and sedimentation (Weiss 1995)..

Wastewater treatment

Findings - Wastewater treatment for residential and commercial lands is divided into two broad groups: decentralized and centralized systems. Decentralized systems are defined as onsite (i.e., individual and large septic systems) and cluster wastewater systems that treat and dispose of relatively small volumes of wastewater, generally from individual or groups of dwellings and business located close together. A centralized system is a collection and treatment system containing collection sewers and a central treatment facility (EPA 1997a). Centralized systems are used to collect and treat large volumes of water. Decentralized systems affect both surface and groundwater, 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 is limited and no national estimates are available. Each state has its own definition of failure, but estimates range from 18 to over 70 percent (EPA 1997a). Twenty-seven states have cited onsite disposal systems as a potential source of groundwater contamination (EPA 1998c). Contaminants from onsite disposal can be categorized into three major groups: inorganic (e.g., sodium, chlorides, potassium, calcium, magnesium, sulfates, and ammonium), microorganisms (e.g., bacteria and viruses) and chemical organics originating in household products (EPA 1997a, Phillips and Lewis 1995). Septic system effluent usually contains high concentrations of ammonium and organic nitrogen. An optimally functioning system maximizes nitrification and minimizes denitrification. Nitrate that is denitrified generally will not be absorbed within the soil and percolates to groundwater.

Water supplies are vulnerable to pathogenic bacteria and viruses from onsite disposal systems. With the exception of Milwaukee, Wisconsin, where 404,000 fell ill to a *Cryptosporidum* outbreak, reported outbreaks of waterborne disease in the United States are uncommon (Table 8.3). However, low occurrence may be attributed to individuals being unaware their illness was a waterborne disease or the number of illnesses were so small that they go unreported by local health departments. Groundwater sources have a higher incident of waterborne outbreaks than surface water (Table 8.4). Disease causing microorganisms isolated from domestic sewage include *Salmonella*, *Shigella*, pseudomonas, fecal coliform and protozoa (*Giardia lamblia*) (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 (EPA 1997b) (Table 8.5).

The fate and transport of parasites, bacteria and viruses from sewage effluent depends on the characteristics of the subsurface environment (EPA 1997a). Pore size and chemical charges of the soil matrix are important in removing bacteria and viruses. Bacteria have been reportedly to travel distances of up to 100 meters in sandy aquifers, 800 m in gravelly aquifers, and 1000 m in limestone rock (Kaplan 1991). Certain viruses, because of their size and long survival times, can travel great distances (i.e., up to one mile) in areas with karst geology (Yates and Yates 1989).

Organic chemicals in onsite disposal systems are a less commonly reported problem because they are often below levels considered hazardous to human health (EPA 1997a), but the can significant affects aquatic systems. Organic chemicals commonly found in septic systems originate from household products, paints/varnishes, shampoos, cosmetics, and polishes.

Septic systems fail for two reasons: poor design or poor maintenance (Kelley and Phillips 1995). Design includes its construction, soils and hydrological characteristics of the site, and drain field layout (Kelley and Phillips 1995). If drainage is too slow, there will be upward seepage and ponding, thus creating a condition for surface water contamination. If drainage is too fast, downward percolation occurs without sufficient biological treatment, thus creating a condition for groundwater contamination.

Maintenance is absolutely necessary even with properly installed systems. Unfortunately, the typical onsite disposal system owner often is unaware of the need for proper maintenance (Kelley and Phillips 1995) and they malfunction.

Class V injections wells include sumps, septic systems, cesspools, and drain fields serving more than 20 persons and can broadly include basically any man-made hole in the ground for injection of wastewater (EPA 1996). They are used by dry cleaners, laundromats, paint dealers, hardware stores, funeral homes and other industrial and commercial facilities for wastes other than domestic and sanitary. Three types of injection wells have high risk for groundwater contamination: motor vehicle waste disposal wells, industrial waste disposal wells and large-capacity cesspools.

Urban Runoff

Findings - Urban lands are a major source of non-point source pollution. People apply various chemicals around their homes, businesses and adjacent land, which are carried by surface runoff to receiving waters. As land is developed, the amount of pollutants discharged into streams increases (Phillips and Lewis 1995). The Nationwide Urban Runoff Program (EPA 1983) reported that, annually, suspended solids discharged from storm sewers serving residential and commercial areas was approximately ten times more than discharged from sewage treatment plants receiving 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), polynuclear aromatic hydrocarbons, and petroleum hydrocarbons (EPA 1997a,Weiss 1995).

Because residential and commercial construction is a major site disturbance, it is highlighted here. Sediment loading from site preparation, construction and maintenance of buildings and roads can exceed the capacity of streams to transport it (Yoder 1995). Sediment loads from uncontrolled or inadequately controlled construction sites typically are 10 to 20 times that of agricultural lands and

1,000 to 2,000 times that of forestlands (Weiss 1995). In a relatively short period, urban site disturbances can contribute more sediment to a stream than previously deposited over several decades (Weiss 1995).

Urban runoff is highly intermittent. Short-term loading, associated with individual events, will be high and may have a shock effect on the receiving waters (Weiss 1995). When predicting the effect of urban run-off on water quality, it is important to determine the duration of the effect. Loading includes an acute (short-term) or a chronic (long-term) effect (Phillips and Lewis 1995). Oxygen-demanding substances and bacteria are examples of an acute effect, whereas nutrients, sediments, toxic metals, and organics are examples of a chronic effect. For an acute effect, estimates are based on the probability that pollution concentrations will exceed acceptable drinking water standard (Phillips and Lewis 1995). For chronic effect, prediction of the increase in pollution loading above current conditions can be estimated using the Simple Method (EPA 1998a). This method uses existing information but is limited to areas less than one square mile.

Ability to Address Issues - Pollution and sediment loading can be mitigated effectively, if based on scientific findings and research. Mitigative practices are to address different aspect of storm management: runoff attenuation, runoff conveyance, runoff pretreatment, and runoff treatment (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). Like plans and guides for wastewater treatment, managers need to check with local and state agencies for specific performance ratings and regulations. Like wastewater treatment facilities, new mitigation practices must be maintained and existing ones upgraded. Adequate funds and updated regulations often are lacking to maintain or enhance these facilities to meet expected performance standards over the long-term (EPA 1997a). Subsequently, water quality may degrade as these units fail.

Underground Storage Tanks

Findings - Within in-holdings, underground storage tanks (USTs) are associated with a small set of facilities. Nearly all USTs contain petroleum products and including service stations and convenience stores and those who have tanks for their own needs (service fleets, local government, federal agencies) (EPA 1998a).

The primary concern of USTs is their leakage can seep into the soil and contaminate groundwater. Since 1988, over 330,000 confirmed releases have occurred from regulated USTs. Although not all of those releases contaminated groundwater, drinking water wells have been shut down because of petroleum contamination (EPA 1996). In 1988, EPA regulations established minimum standards for new tanks and required owners to upgrade existing tanks, to replace or close them by December 1998 (EPA 1996). Current estimates indicate that 25 to 35% of USTs have not met compliance.

Recent, studies have identified methyl tertiary butyl ester (MTBE) to be potentially a major health hazard in drinking water. MTBE is added to gasoline to increase its oxygen content and to reduce certain emissions. Effects on drinking water resources include widespread impacts occurring at low concentrations and local impacts occurring at high concentrations (EPA 1998b). Widespread impacts result from vehicular emissions that dissolve in rain or snowfall and subsequently infiltrate to shallow groundwater..

Local impacts primarily result from leaking USTs. A survey of groundwater plume data from over 700 service stations identified that 43% of the sites had MTBE concentrations greater than 1,000 µg/L. However, a survey of drinking water wells from 20 NAWQA study units showed that 2% of 949 rural wells had a median concentration of approximately 0.5 µg/L, well be below the EPA Drinking Water Advisory of 20-24 µg/L (EPA 1998b, Zogorski et al. 1998). A study in Maine of private wells showed a higher percent of 1.1% of 951 wells with levels exceeding 35 µg/L and Maine officials estimated that 1,400-5,200 private wells across the state could be contaminated at levels exceeding 35 µg/L (EPA 1998b). More data are needed to determine the extent of contamination of drinking water sources by MTBE and the potential health hazard. With time, the potential threat of USTs contaminating groundwater should diminish as older tanks are upgraded and sites are cleared of contaminates.

Solid Waste Landfills

Introduction - In 1990, citizens in the United States generated over 195 million tons of municipal solid waste. Currently, there are over 6,000 regulated municipal landfills (EPA 1993)specific concern are the potential impacts on water quality and the environment. In 1976, Congress enacted 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 landfill. The practice was to spread hazardous waste sludge and liquids over municipal waste, and using the municipal waste to soak up the waste (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 effect of hazardous waste management on drinking water quality is reviewed in chapter 12. In this section, we discuss the effect of municipal solid waste in landfills on drinking water quality.

EPA (1993) defines a municipal solid waste landfill as follows: “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 and these landfills may be exempt from some of regulation requirements. EPA (1993) defines a small landfill as receiving less than an average of 20 tons of waste per day, receives less than 25 inches of rain per year, and shows no evidence of groundwater contamination. About half of the regulated landfills serve communities less than 10,000 people and are considered small landfills and many of these small landfills are within the urban/wild land intermix.

Findings - Often, those municipal solid waste landfills that contaminated groundwater were poorly designed, located in geologically unsound areas, or accepted toxic materials without proper safeguards (EPA 1993). Decomposition of municipal solid waste in landfills forms a leachate, a liquid containing extremely high concentrations of organic and inorganic pollutants. Groundwater contamination is common near landfills; however, the effect decreases with distance (Borden and Yanoschak 1990). A study of 71 North Carolina sanitary landfills found that 53 percent had groundwater violations for organic and inorganic pollution based on North Carolina Groundwater Quality Standards (Borden and Yanoschak 1990). Organic contamination was localized to just a few landfills.

A comparison of monitoring data for landfills and secondary wastewater treatment plants discharging into surface waters indicated a higher concentration of heavy metals from landfills, but organic pollutants concentrations were similar (Borden and Yanoschak 1990). In a different study, organic pollutants from municipal landfill leachate had similar concentrations to leachate from industrial landfills (Brown and Donnelly 1988). Thus, it is important to know the landfill's age, history of material disposal, design, and its capability of handling toxic waste.

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

Reliability of Findings - Although in-holding development has existed for decades, scientific studies of their effect on water quality and drinking water sources are lacking. Nevertheless, extensive research has been conducted on urban effects on natural systems. These studies provide the basis for identifying the potential impacts of development on drinking water.

Ability to Extrapolate Findings - When applying findings across a watershed, scale becomes an important issue. At the local scale, in-holdings, wastewater treatment, urban runoff, and/or landfills may have minimal effect. However collectively and at the broad-scale, the cumulative effects may be great. Consequently, these developments must be evaluated independently and collectively within the watershed. Age of landfills, wastewater treatment, and urban/storm runoff structures must be considered. Existing structures may or may not meet existing sanitation and water quality regulations for different reasons. Current and future management actions may be predicated on the functioning of that infrastructure.

Secondary linkages - Besides the potential negative effects of in-holding development on drinking water sources, development directly modifies habitat through land conversion from natural vegetation to urban land use. Habitat modification results in the loss of both terrestrial and aquatic habitats, and fragmentation and isolation of remaining habitat. For example, an increase in impervious area of only 10% of the watershed can significantly degrade a stream channel and its biota and riparian function (Klein 1979,Schueler 1994). These modifications can alter numbers, distribution and migration of a species.

In-holdings influence natural resource management by altering practices and costs (Zipperer 1993). Resource managers may need to alter or stop management prescriptions to meet desires of in-holding residents. These changes potentially may alter management objectives and goals for the public land manager. Further, in-holding development may limit access to established trailheads as owners deny passage through their property. Consequently, new trailheads need to be constructed; an action that must compete for funds with other recreation objectives.

Ability to Address Issue – The ability to address the effects of development on drinking water quality is dependent on ownership. For publicly owned lands, resource managers directly determine the compliance of facilities to meet federal and state regulations. However, for privately owned lands, resource managers may indirectly influence development effects on drinking water quality through participating in the planning process.

Development and land-use planning of private lands within and adjacent to public lands are complex issues involving 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 lands are owned by a diverse group of people with diverse reasons for ownership. Third, changes in drinking water regulations and statues create the need for communication and education. Fourth, a comprehensive approach is needed to account for the cumulative effects of individual development within a watershed and to address the need of individual stakeholders.

Because of the interplay, technology and management practices are not the only solution to mitigating 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 land management strategies through integrating and coordinating management priorities across stakeholders, governments and agencies. Livingston (1993) provides key elements for this planning process.

Manuals have been published on wastewater and urban storm water management. These manuals contain guidelines for site conditions such as soil characteristics (permeability, depth to bedrock, depth to groundwater table), topography (flood plain, hill slope, ridge top) and climatic patterns (rain and snow fall amounts and patterns, winter temperatures). Each site needs to be addressed individually when designing, constructing, and maintaining management practices.

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

- Identification of landfill sites: proximity to wells, aquifers, geological and hydrological features, surface waters?
- Knowledge of the landfill age:
    1. Old landfills: Landfills existing before RCRA, may contain hazardous material and may be improperly designed for hazardous material storage and municipal waste.
    2. Existing landfill: Landfills existing after RCRA may still pose a problem for groundwater contamination because the site may contain older units where hazardous waste was deposited, have improper design, and have punctured liners or clay layers.
    3. New landfills: Landfills being managed under current federal and state regulation; however, size (e.g., small landfills) is importance with regards to exemption to regulations.
    - Knowledge of landfill history: For example, what was deposited on the site and when. How was the landfill constructed—clay layer, liner or combination of the two.
    - Monitoring data: Is the site being monitored for VOCs and groundwater contamination? Is monitoring sufficient to safeguard groundwater sources?
    - Extent of contamination plume. If groundwater is contaminated, what is the vertical and horizontal extent of the contamination? What is the effect of the plume on drinking water sources?
    - Compliance with current state and federal regulations: What mitigation actions have been taken to comply with state and federal laws if contamination occurred.

Since the need for landfills exits, the design and management of safe landfills is paramount. To meet this need, federal, state, Indian tribal and local governments have adapted an integrative approach that involves three waste management techniques: decrease the amount through source reduction, increase recycling of materials, and improve design and management of landfills (EPA 1993). Because a number of regulations exist for the management of a municipal solid waste landfill and the flexibility within the regulations to meet local conditions, managers are recommended to contact a local EPA or state natural resource agency office for information on siting, design and management for their area.

Research needs – Development of in-holdings represents an opportunity to examine how development alters ecosystem processes and what are the long-term implications of these changes. Long-term monitoring stations are needed not only to monitor changes in water quality and habitat modification but also atmospheric deposition. With increased development, a corresponding increase in vehicular emissions will occur. These emissions (e.g., MTBE) may have negative consequences on surface and groundwater quality and on soil biota and processes. In addition, studies are needed to determine the limitation of management practices for landfills, wastewater treatment, and urban runoff for mountainous environments and at high elevations (>8,000 ft.). Research is needed to determine threshold levels of change in biodiversity, biogeochemical processes.

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Table 8.1. Reported acres by region within Nation Forest boundary and percent population change of counties containing or adjacent to a National Forest. Population changes are reported only for those forests within the conterminous United States.

Region *	Gross Acreage (acres)	NF Acreage (acres)	In-holding Acreage (acres)	Other Acreage (%)	Population Change (%) (1980-1990)	Population Change (%) (1990-1996)
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
10	24,355,135	22,953,445	2,301,690	9.5		
Total	231,719,065	191,601835	40,117,230	17.3		

*Region 1: Montana, northern Idaho, North Dakota, and northwestern South Dakota
Region 2: Colorado, Kansas, Nebraska, and southeastern South Dakota
Region 3: Arizona, 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, Washington
Region 8: Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, Texas and Virginia
Region 9: Connecticut, Delaware, Illinois, Indiana, Iowa, Maine, Maryland, Massachusetts, Michigan, Minnesota, Missouri, New Hampshire, New Jersey, New York, Ohio, Pennsylvania, Puerto Rico, Rhode Island, Vermont, West Virginia, Virgin Islands, Wisconsin

Table 8.2. Summary of estimated use (cubic kilometers) from freshwater surface and groundwater sources in the United States from 1980 to 1995 (adapted from Gleick 1999).

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

Table 8.3. Waterborne disease outbreaks in the United States by water supply system from 1990-1994 (adapted from Gleick 1999).

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[1]	1,176

[1]Includes Milwaukee, Wisconsin

Table 8.4. Comparison of outbreak percentages by drinking water sources from pathogenic contamination for the period 1971-1996 (adapted from Craun and Calderon 1996,EPA)[1].

Water Source	Total outbreaks	Cases of Illnesses
Ground	371 (58%)	84,408 (52%)
Surface	215 (33%)	66,721 (41%)
Other	56 (9%)	10,625 (7%)

[1]Excludes outbreak in Milwaukee, Wisconsin, 1993

Table 8.5. Pathogenic bacteria and viruses identified in drinking water in the United States (adapted from EPA 1999).

Waterborne pathogenic bacteria	Waterborne pathogenic viruses
Legionella	Enteroviruses
Mycabacterium avium intracellular (MAC)	Coxsackieviruses
Shigella (several strains)	Echoviruses
Heliobacter pylori	Poliovirus
Vibrio cholerae	Enteroviruses
Salomonella typhi	Hepatitis A virus
Salomonella typhimurum	Hepatitis E virus
Yersinia	Enteric Adenoviruses
Campylobacter (several strains)	Rotavirus
Esherichia coli (several strains)	Norwalk virus
	Small round structured viruses
	Astrovirus
	Caliciviruses

Chapter 9

Mining, Oil, and Gas Extraction

Mineral, oil, and gas operations occur in watershed supplying drinking water. These activities can have significant and longlasting impacts on water quality.

Potential Water Quality Impacts Of HardRock Mining

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Introduction – Mining can significantly impact water quality for domestic water supplies. These impacts can be long term, vary by mining method, method of ore processing, method of water management, and method of mine closure. The impacts include sediment, acid mine drainage, chemical, and toxic metals.

The focus of this section is hardrock mining, not including coal mining. Coal mines can have significant water quality impacts and are regulated under a specific set of federal and state laws and programs.

Hardrock mining is defined as the extraction of precious and industrial metals and non-fuel minerals by surface and underground mining methods (Lyon, et,al). Domestic mining of metals and minerals has been an important industry in the United States for more than a century and has occurred in almost every state. Many metals and minerals are valuable resources and an economic necessity in many states. Extensive mining started in the 1880s and for the next 70 to 80 years, hardrock mining was a major industry in many states. The legacy of this active period is 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.

Hardrock mining is a large-scale activity that typically disturbs large areas of land. The siting of a mine is dictated by the location of the ore body. Many ore bodies and mines are located on public land administered by the Federal land management agencies, i.e., USDA-USFS and USDOI agencies. Mines 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. Environmental controls were very limited or non-existent during first half of the century and abandoned mines are causing serious environmental damage.

Because of the high waste-to-product ratios associated with mining most ore bodies, large volumes of mining related waste is generated. Mine waste includes material generated as a result of mining and ore processing activities. Most waste materials are considered to be non-marketable. However, mine waste materials often contain environmentally significant concentrations of heavy metals and precious metals. The intent of this 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. The method used depends on the type of metal or mineral being mined, the nature of the ore body and cost. Surface mining or open pit methods are typically used for large shallow ore bodies, which have a low metal or mineral value per volume of rock. Underground mining methods are typically used when the mineralized rock is deep and occurs in veins.

Surface or open pit mining removes and deposits surface soil and rock overburden, which contain no target mineral. The underlying ore body typically includes rock containing uneconomical concentrations of the target mineral, which is removed and typically stockpiled or otherwise disposed 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. Once the ore body is reached horizontal passages called drifts and crosscuts are developed on numerous levels and the ore is mined. Blasted ore and waste rock are transported to the surface via rails, small trucks or hoisted to the surface in vertical shafts.

Ore Extraction

In both surface and underground mining, extraction of ore waste materials requires the use of heavy equipment and explosives. Once at the surface the rock and ore is typically transferred to larger trucks for transportation to storage or processing facilities. Overburden and waste rock usually do not contain minerals that may impact surface or ground water. Waste rock can be used as mine backfill, however most of the waste rock generated is disposed of in piles near the mine site.

Surface and underground mines typically extend below the local and/or regional water table. This results in ground water flowing into the mine pit or underground workings. Deep mines can intersect deeper confined aquifers which provides a conduit for this water to move upward towards the land surface. 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 level to allow for free drainage of the water entering the upper workings.

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. 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. The mercury coated gold particles will 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 and others sink. The waste (tailings) and the wastewater are typically disposed of in large, constructed impoundments.

Leaching - Leaching refers to processes that involve pouring an acid or cyanide solution over crushed and uncrushed ore to dissolve metals into solution 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 uses a nearby holding area (typically a pond) to store the "pregnant" solution prior to recovery of the desired mineral using chemical or electrical processes. Once the desired mineral is recovered the solution is recycled 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 and contain millions of metric tons of ore rock. Leaching solutions aided by precipitation dissolves the desired minerals into solution. 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 is lost to the subsurface, reduces the amount transported to the holding pond, and creates potential environmental problems. Dump leaching is used for very low grade ore (copper - 0.05 percent or more).

Heap leaching is used for high grade ores (copper - 0.5 to 1 percent) 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 may recover economic quantities of the desired mineral for months, years or decades. When leaching no longer produces economically attractive 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.

Common reagents used include copper, zinc, chromium, cyanide, nitrate and phenolic compounds, and, at copper leaching operations, sulfuric acid. For non-leaching operations, the quantities of reagents used are very small compared with the volumes of water generated, thus, the risks of toxic pollutants releases limited.

Water management

Management of water at large mine sites is a critical. Water management is difficult and complicated by the multiple needs. The needs include draining open pits and/or underground workings, transporting surface runoff across the site, using water in ore processing, and meeting water quality standards.

Waste management

Hardrock mining typically produces large volumes of solid waste types including overburden, waste rock, spent ore, and tailings. Overburden and waste rock are typically stockpiled at the mine site, used as backfill, and usually pose limited threat to the environment. Some 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 and disposed of in tailing ponds. Modern tailing impoundment design includes low permeability 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. Impoundment seepage raises the probability of surface water and ground water contamination and, coupled with the potential for acid rock runoff, may require long-term water treatment well after the active life of the facility. Impoundment structures can fail, releasing tailings and contained mill effluent.

Spent ore is a waste material that is generated at mines that utilize dump or heap leach processes. 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 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, the nature of the area, and the ore process used, active management of the mine site and water management may be necessary for years or even decades closure. Until recently, reclamation was limited to grading and re-vegetating waste materials and pits to minimize erosion and improve the visual landscape. Permanent closure now includes: removal/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, active water management, and assuring applicable water quality standards are met. In numerous cases, this has meant operating and maintaining a water treatment facility. At mine sites where acid mine drainage is a problem, post closure water treatment may be necessary for decades. Other environmental controls including roads, water ditches and diversion structures, siltation ponds, spillways, dam stability and treatment sludge management require significant post closure investment

The long-term nature of mining impacts requires that environmental monitoring (including compliance monitoring), contingency planning and financial assurance be effective for many decades. Geochemical conditions in waste rock, tailings and underground workings can change over time. This requires the ability to make necessary changes in water control and water treatment facilities after mine closure.

Issue and Risks - Facilities associated with hardrock mines - i.e., mine adits and shafts, underground workings, open pits, 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 high concentrations, these contaminants can have significant negative affects on the quantity, quality and usability of surface water and ground water. Dissolved and total metals concentrations in source waters can impact the use of water for public water supply and the aquatic health of stream and riparian systems.

The mobility of the contaminants is magnified by exposure to rainfall and snowfall. Surface runoff is a key mechanism by which pollutants are released into surface waters. Seepage from tailing ponds and waste rock piles, unwanted releases from process water ponds or waste water ponds, drainage from underground workings, and discharge of pit water are other means of contamination. Surface waters may be impacted indirectly via ground water. Impacts to surface waters include sediments and sediments contaminated with heavy metals.

A variety of complex geochemical and hydrogeological processes control the transport, attenuation and ultimate distribution of heavy metals and other mining-related contaminants in surface and ground waters. Dissolved and suspended contaminants are transported to aquifers and streams via complex subsurface and overland flow pathways. This complexity, combined with the large scale of mining activities and numerous types of mine facilities, makes the water quality assessments, and restoration and remedying pollution very difficult.

Findings - The major types of water quality impacts from mining are discussed briefly below. The following sections discuss erosion and sedimentation, acid rock drainage, cyanide leaching and dissolution, and transport of toxic metals.

Erosion and Sedimentation.

Because large areas of land are disturbed by mining operations and the large quantities of earthen materials are exposed at sites, erosion can be a major concern. Major mining sources of erosion and sediment include open pit areas, heap and dump leaches, waste rock and overburden piles, tailings piles and dams, haul roads and access roads, ore stockpiles, vehicle and equipment maintenance areas, exploration areas, and reclamation areas. Historically, erosion and sediment from mining have built up of thick layers of mineral fines and sediment in flood plains and streams. Consequently, erosion and sediment control must be considered from the beginning of operations through post closure operations. Of special concern, sediment can carry attached chemical pollutants and toxic metals,

The types of impacts are numerous, producing both short-term and long-term impacts. Impacts include:

1. chronic and acute toxic effect in fish,
2. destroying aquatic habitat for fish and benthic macroinvertebrates,
3. chemicals and toxic metals posing threat to human health through drinking water,
4. chemicals and toxic metals bioaccumulating in fish and consumed by humans, and
5. increase stream turbidity.

Sediments and minerals deposited in flood plains can impact the quality of nearby surface water bodies and underlying ground water. Minerals may lower the pH of surface runoff thereby mobilizing heavy metals that can infiltrate into underlying sediments or be carried to nearby surface waters. Lower soil pH may cause the loss of riparian vegetation.

In addition, mining increases streamflow velocities and volumes, downstream flooding, scouring of stream channels, and structural damage to water diversions, drinking water intakes, bridge footings, and culverts. These impacts can increase sediment loads and turbidity of streams.

Acid Mine Drainage

A major water quality problem is the formation of acid mine drainage (AMD) and the associated mobilization of toxic metals and sulfate. The formation of AMD results from the exposure of sulfide minerals (typically pyrite) to air and water. Pyrite and other sulfide minerals (pyrrhotite, galena, sphalerite and chalcopyrite) are associated with coal deposits, precious and heavy metal ore bodies, waste rock and process wastes (tailings). Acid drainage can discharge from underground mine workings, open pit walls and floors, tailings impoundments, the toes of waste rock piles and spent ore from heap and dump leach operations. The most common metals associated with sulfide rocks include lead, zinc, copper, arsenic, cobalt, molybdenum and antimony. Acid drainage is generated at both abandoned and active mine sites.

Pyrite reacts with water and oxygen to produce ferrous iron, sulfate and acid (H+). In oxidizing conditions and at a pH greater than 3.5 the ferrous iron will oxydize to ferric iron - Fe+3.
Much of the ferric iron will then precipitate as iron oxide - Fe(OH)3. Iron oxide is typically referred to as "yellowboy". The ferric iron remains in solution and continues to attack the pyrite surface and accelerate the oxidation of the pyrite and subsequent production of acid. As pH decreases the precipitation of ferric hydroxide decreases resulting in greater dissolved concentrations of ferric iron and therefore a greater rate of sulfide oxidation. To make matters worse the sulfide oxidation rate is increased by 5 or 6 orders of magnitude by the Thiobacillus ferrooxidans bacterium. These bacterium, which catalyze the oxidation reaction, are very common in the subsurface as they can grow in the absence of light and with minimal oxygen.

Both water and oxygen are necessary to generate acid drainage. Water serves as both a reactant and a medium for bacteria to catalyze the oxidation process. Water also transports the oxidation products. A ready supply of atmospheric oxygen is required to drive the oxidation reaction. Oxygen is particularly important to maintain the rapid bacterially catalyzed oxidation at pH values below 3.5. Oxidation is significantly reduced when the concentration of oxygen in the pore space is less than 1 or 2 percent. AMD commonly contains high concentrations of TDS, sulfate, iron and other metals, and has low pH values.

The acid water will dissolve carbonate minerals such as calcite. As acid water encounters carbonate minerals, the calcium is released and bonds with the sulfate to form gypsum which precipitates.

Acid generation and drainage affect both surface and ground water, thus aquatic birds, fish and other aquatic organisms, and humans. Ingestion of contaminated surface water or direct contact through outdoor activities such as swimming can affect humans. The release of metals and other mine related contaminants to sources of drinking water can cause standards to be exceeded, thus requiring expensive treatment or acquisition of another source of water by a public water supply.

Cyanide Heap Leaching

For over a century, cyanide has been used as a pyrite depressant in base metal flotation and in gold extraction. Continued improvements in cyanide leaching technology have allowed the economic mining of increasingly lower-grade gold ores. With continued high gold prices, these improvements have resulted in increasing amounts of cyanide being used in mining. The mining industry now uses must of the sodium cyanide produced in the United States, with more than 100 million pounds used by gold/silver leaching operations in 1990. Dump leaching and heap leaching operations commonly utilize a cyanide solution as the leaching solution.

The use of cyanide can cause three major types of environmental impacts: first, cyanide-containing ponds and ditches can present an acute hazard to wildlife and birds. Tailings ponds present similar hazards, but less frequently because of lower cyanide concentrations. Second, spills can result in cyanide reaching surface water or ground water and cause short-term (e.g., fish kills) or long-term (e.g., contamination of drinking water) impacts. Finally, cyanide in active heaps, ponds, mining wastes, spent ore heaps, dumps and tailings impoundments may be released and present hazards to surface water or ground water. Geochemical changes can affect the mobility of heavy metals.

The acute toxicity of cyanide, and many major incidents, have focused attention on the use of cyanide in the mining industry. When exposure occurs by inhalation or ingestion, cyanide interferes with many organisms' oxygen metabolism and can be lethal in a short time. Other contaminants associated with cyanide solutions include nitrate, which is a cyanide breakdown product, and dissolved heavy metals. Cyanide is much more toxic to aquatic organisms than to humans. The acute aquatic standard is 22 ug/l and the chronic aquatic standard is 5.2 ug/l. These values are for total cyanide and depend on hardness. The Maximum Contaminant Level for public drinking water supplies is 200 ug/l.

In arid regions, with limited water resources, the amount of water necessary to rinse heaps to a required standard can be a significant concern. Conversely in wet climates, excess water from heavy precipitation can place a strain on system operations and may make draining or re-vegetating a heap or impoundment very difficult. In addition, the chemistry of a spent heap or tailings impoundment may change over time. Although effluent samples at closure/reclamation may meet state requirements, the effluent characteristics may be dependent on the pH. Factors affecting chemical changes in a heap or tailings impoundment include pH, moisture, mobility, and geochemical stability of the material. The principal concerns with the closure of spent ore and tailing impoundments are long-term structural stability and potential to continue leach contaminants. The physical characteristics of the waste material (e.g., percent slimes vs. sands in impoundments), the physical configuration of the waste unit, and site conditions (e.g., timing and nature of precipitation, upstream/uphill area that will provide inflows) influence structural stability.

Transport of Dissolved Contaminants

Dissolved contaminants (primarily metals, sulfates, and nitrates) can migrate from mining operations to local ground and surface water resources which are utilized for public water supply. While the low pH associated with AMD can enhance contaminant mobility by promoting metals leaching from mine workings and mine wastes, releases can also occur under neutral pH conditions. Discharges of process water, mine water, runoff, and seepage are the primary transport mechanisms to surface water and ground water. In addition naturally occurring substances in the ore can be a significant source of contaminants. Mined ore not only contains the mineral being extracted but varying concentrations of a wide range of other minerals, including radioactive minerals. Frequently other minerals may be present at much higher concentrations and can be much more mobile than the target mineral. Depending on the local geology, the ore, the surrounding waste rock and overburden can include trace levels of aluminum, arsenic, asbestos, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, silver, selenium, and zinc, as well as naturally occurring radioactive materials.

Under specific conditions dissolved pollutants discharged to surface waters can attach to sediments. Specifically, some toxic constituents including lead and mercury are often found at elevated levels in sediments, while being undetected in the water column. Sediment contamination may affect human health through consumption of fish that bioaccumulate toxic pollutants. Furthermore, elevated levels of toxic pollutants in sediments can have direct acute and chronic impacts on macroinvertebrates and other aquatic life. Finally, contaminated sediment provides a long-term potential source of pollutants if dissolved in the water column.

The ability of pollutants to dissolve and migrate from materials or workings to ground water varies significantly depending on the constituent of concern, the nature of the material/waste, the design of the management, soil characteristics, and local hydrogeology including depth, flows, and geochemistry of the underlying aquifers. Risks to human health and the environment from contaminated ground water usage vary with the types of and distance to local users. In addition, impacts on ground water can also indirectly affect surface water quality.

Zinc and other base and precious metals were produced from ores excavated from an underground mine in central Colorado from 1878 to 1977. The resultant wastes consist of roaster piles, tailing ponds, piles of waste rock and acid drainage from the mine. Percolation from the tailing ponds has contaminated ground water below and down gradient of the ponds. The ground water discharges to a nearby stream. Runoff from the roaster and waste piles and acid drainage from the mine discharge directly to the stream. The main pollutants are pH, arsenic, cadmium, copper, lead, manganese, nickel, and zinc. Concentrations of cadmium, copper, and zinc exceed water quality criteria in the stream. In addition, levels of dissolved solids are above background concentrations. At least two private wells previously used for drinking water have been contaminated.

Ability to Extrapolate Findings - The information and conclusions presented above apply to the extraction and ore processing of precious, industrial and non-fuel minerals.

Secondary Linkages - The contamination of drinking water resources due to mining operations is linked to aquatic impacts, hydrologic impacts, ground water drawdown, subsidence, and soil contamination.

Hydrologic Impacts

Mining operations interact with the site hydrology. Mine design, and location, can affect the hydrologic setting in a number of ways:

1. Regional surface and ground water movement.
2. Groundwater flowing into the mine and subsequent contact with mining relate pollutants.
3. Surface water inflow and precipitation related recharge.
4. Increases in surface and ground water interaction with the mine workings because of subsidence.
5. Loss of surface features, such as lakes through subsidence.
6. Pathways for post closure flow resulting from adits, shafts, and overall mine design.
7. Operational and post closure geochemistry and resulting toxics mobility.
8. Overall site water and mass balance.

Ground Water Drawdown.

Ground water drawdown and associated impacts to surface waters and nearby wetlands can be a serious concern in some areas:

1. reduction or elimination of surface water flows,
2. reduce or eliminate seeps and springs.
3. degradation of surface water quality and beneficial uses,
4. degrade or eliminate associated wetland areas.
5. degradation of habitat associated with springs and seeps,
6. upland habitats as ground water levels decline below the deep root zone,
7. reduced or eliminated production in domestic supply wells, and
8. sediment and other water quality/quantity problems associated with discharge of the pumped water back into surface waters.
9. The impacts could last for many decades.

While dewatering is occurring, discharge of the pumped water with appropriate treatment, can be used to mitigate adverse effects on surface waters. However, when dewatering ceases, the cones of depression may take many decades to recharge and may continue to reduce surface flows in rivers and tributaries.

Subsidence

Mining subsidence occurs when overlying strata collapse into mine voids. The potential for subsidence exists for all forms of underground mining. Subsidence may manifest itself as sinkholes or troughs. Sinkholes are usually associated with the collapse of part of a mine void. The extent of sinkholes are usually limited in size. Subsidence of large portions of the underground void forms troughs, typically over areas where most of the resource had been removed. The threat and extent of subsidence is related to the method of mining employed. Typically, traditional room and pillar methods leave enough material in place to avoid subsidence effects. However, high-volume extraction techniques, such as pillar retreat, can increase the likelihood that subsidence will occur. At some mines, waste rock and/or stabilized tailings are backfilled in the mine to minimize subsidence.

Effects of subsidence may not be confined to or even visible from the ground surface. Sinkholes or depressions in the landscape interrupt surface water drainage patterns; ponds and streams may be drained or channels may be redirected. Irrigation systems and drainage may be disrupted. In developed areas, subsidence has the potential to affect building foundations and walls, highways, and pipelines. Ground water flow may be interrupted or disrupted as impermeable strata break down, and this could result in flooding of the mine voids. Impacts to ground water include changes in water quality and flow patterns, including surface water recharge.

Soil Contamination

Human health and environmental risks from soils generally fall into two categories: (1) contaminated soil resulting from windblown dust and stack emissions, and (2) soils contaminated from chemical spills and residues. Fugitive dust can pose significant environmental problems at some mines. The inherent toxicity of the dust depends upon the proximity of environmental receptors and type of ore being mined. High levels of arsenic, lead, and radionuclides in windblown dust usually pose the greatest risk. Soils contaminated from chemical spills and residues may pose a direct contact risk when these materials are misused as fill materials, ornamental landscaping, or soil supplements.

Ability to Address Issues - Management practices are used to control erosion and sediment 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, yet need year-round controls because contaminant sources remain exposed to precipitation. At least six categories of management practice options are available to limit erosion and the off-site transport of sediment, including discharge diversions; drainage/storm water conveyance systems; runoff dispersion; sediment control and collection; vegetation and soil stabilization; and capping of contaminated sources.

No easy or inexpensive solutions to acid drainage exist. Two primary approaches to addressing acid generation are 1) avoiding mining deposits with high acid generating potential and 2) isolating or otherwise special-handling wastes with acid generation potential. In practice, avoiding mining in areas with the potential to generate acids is difficult, if not impossible. Isolation of materials with the potential to generate acids is now being tried as a means of reducing the perpetual effects to surface water and ground water from mining wastes. Isolating materials can be implemented by preventing or minimizing oxygen contact with the material, preventing water from contacting the material, and/or ensuring that an adequate amount of natural or introduced material is available to neutralize any acid produced. Techniques used to isolate acid generating materials include subaqueous disposal, covers, waste blending, hydrologic controls, bacterial control, and treatment.

Acid generation prediction tests are increasingly relied upon to assess the long-term potential of a material, or waste, to generate acid. Mineralogy and other factors affecting the potential for AMD formation are highly variable from site to site, and this can result in difficult, costly, and questionable predictions. In general, the methods used to predict the acid generation potential are classified as either static or kinetic. Static tests are not intended to predict the rate of acid generation, only the potential to produce acid. Static tests can be conducted quickly and are inexpensive compared with kinetic tests. Kinetic tests are intended to mimic the processes found in the waste unit environment, usually at an accelerated rate. These tests require more time and are much more expensive than static tests.

The heightened awareness of the potential environmental problems associated with cyanide leaching have led federal land managers and states to develop and implement increasingly stringent regulations or, more often, non-mandatory guidelines. These regulations and/or guidelines address the design of facilities that use cyanide. These regulations and guidelines include requiring or recommending liners and site preparation for heap leach piles or tailings impoundments, and address operational concerns of monitoring of solutions in processes and in ponds, and sometimes treatment requirements for cyanide-containing wastes, and closure and reclamation requirements. Operators are generally required to take steps either to reduce or eliminate access to cyanide solutions or to reduce cyanide concentrations in exposed materials to below lethal levels. Regulatory requirements and guidelines as to the allowable concentration of cyanide in exposed process solutions are widely variable (when numeric limitations are established, they generally range around 50 mg/l), as are the means by which operators comply. Operators reduce access in several ways, including covering solution ponds with netting or covers, using cannons and other hazing devices (e.g., decoy owls) to scare off waterfowl and other wildlife, and/or installing fencing to preclude access by large wildlife.

Because hardrock mining can have major impacts on water quality, in-stream monitoring of mines and ore processing facilities is needed to assess impacts, to alert watershed managers and water quality agencies to problems, and to address problems.

Research Needs –

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Oil and Gas Development

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Introduction - Oil and gas exploration is characterized as having generally short-term effects. It consists of geologic mapping and ground geophysical methods consisting of surface gravity, magnetic, and seismic surveys of the prospective area. Gravity and magnetic data is 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. More likely, today the seismic energy is generated by thumpers mounted on large trucks, utilizing less environmentally sensitive 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.

Exploratory well drilling entails both site occupancy and reconfiguration. It has relatively short-term effects. Exploratory wells afford an opportunity to acquire drill cuttings and core for visual analysis, as well as an opportunity to geophysically 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 by rotary drilling requiring fluid circulation to lubricate and cool the bit, prevent plugging of the hole, maintain the necessary hydrostatic pressure to prevent collapse of the well, and counterbalance any high pressure oil, gas, or water encountered in any of the drilled formations, thus aiding in the prevention of a blowout, a catastrophic surge of highly pressurized fluid into the wellbore that can cause fires and loss of life and property.

The fluid circulation system is generally referred to as drilling mud. Drilling muds are primarily water-based mixtures of clays and inert weighting constituents 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 a 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, the bit, and back up the annular space between the drill stem and the hole. Lined or unlined pits are used to store supplies of water, waste fluids from drilling, rock cuttings, rigwash, and stormwater runoff. Lined pits are normally used for saltwater or oil-based mud and unlined pits are normally used for freshwater mud systems The mud is screened and filtered and recirculated through tanks back into the hole.

After the first few hundred feet of well are drilled, casing is set. Upon reaching the desired depth, the well is analyzed by electric and nuclear logs to determine whether the hole is a potential producer. If it is, the production casing is cemented into the wellbore and the drilling rig is replaced by 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 foreign fluids from mixing with the producing formation; and it isolates the producing zone or other contaminating zones in the well from contact with freshwater acquifers.

Once drilled, cased and completed, many wells have insufficient force to flow without further assistance because of contamination by foreign material introduced by drilling or by indigenous material within the formation. 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 the reservoir is so low that it is difficult for the oil and gas to flow into the well, the rock must 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 by pump trucks to fracture the formation.

In the production phase of oil and gas development, the drill is removed from the site and replaced by the well head. It has long-term effects because the facilities associated with it are in place over the life of the reservoir. A flowing well is any well which has sufficient pressure in the reservoir 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 from the reservoir to the surface. Depending upon the particular circumstances, 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 the natural reservoir energy available, primary recovery can range from less than five percent to 75% 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.

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. Generally, oil produced from the well is a mixture of oil, water, gas, and sand or other solid material. The sand and other solid material 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. Heater treaters 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 the heater treater process.

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. Typically, they have a capacity for two to three days of production. Facilities typically include provisions for transfer to trucks or transfer to pipeline.

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 produce well and primary recovery methods are usually sufficient. Recovery is often greater than 80% of the resource.

Issues and risks - In Section 3001 of the 1980 amendments to the Resource Conservation and Recovery Act (RCRA), Congress 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 Environmental Protection Agency (EPA), examples of the exempted associated wastes include: well completion, treatment, and stimulation fluids; basic tank sediment and water from storage facilities that hold product or exempt waste; workover wastes; packing fluids; and constituents removed from produced water before it is injected or disposed of otherwise. Some associated wastes not believed by EPA to be exempt include: unused fracturing fluids or acids; gas plant cooling tower cleaning wastes; oil and gas service company wastes, spent solvents, spilled chemicals, and waste acids.

The access roads and well pads erode and are sources of sediment during the exploration and production phases (See Chapter 11 for discussion of roads and sediment).

Under the Clean Water Act, discharges to surface waters by oil and gas exploration and production activities are addressed by the National Pollutant Discharge Elimination System. On-shore discharges, including potentially contaminated runoff, are prohibited except from wells producing not more than 10 barrels per well per day and discharges of produced water that are determined to be beneficial to agriculture or wildlife.

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 groundwater 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 (USDWs). A USDW is an aquifer which supplies drinking water for human consumption or for any public water system, or contains fewer than 10,000 mg per liter of total dissolved solids, and 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 water 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 used to dispose of produced waters, a byproduct of oil and gas recovery, or to inject fluids to enhance production. Most produced water is strongly saline, with total dissolved solids ranging from several hundred parts per million (ppm) to over 150,000 ppm. Because up to 70% of the total recoverable oil may remain in a producing formation during primary recovery pumping, additional energy must be supplied to the reservoir by the controlled injection of water from the surface through injection wells to achieve secondary recovery. 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 usable aquifers can occur as a result of improperly plugged abandoned wells, casings, and through direct injection into aquifers. During exploratory and development drilling, the well has the potential to act as a conduit between formations containing hydrocarbons, heavy metals or chlorides associated with accompanying brines and formations providing usable acquifers. 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 aquifer.

Stimulation of a reservoir utilizing the pumping of a fracture fluid under high pressure into the formation can have adverse effects. If the fractures extend beyond the boundaries of the reservoir and encroach a freshwater aquifer, the fracture fluids may migrate into the aquifer.

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 brine water in old injection wells with leaking casings can inject brine water into surface geologic strata which can percolate to and contaminate surface waters, too. Sometimes brine water is trucked to injections wells, however some truckers have been known to dump the brine 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 and runoff to streams. These pipes cross streams, can break, and discharge directly into the stream. The degree of contamination depends upon, among other things, the extent and duration of the leakage.

Some waste management practices associated with hydrocarbon production may have an effect on groundwater. 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, if present. Some natural gas contains hydrogen sulfide, carbon dioxide, or other impurities that must be removed prior to sale. Sweetening is the removal of these impurities by various chemical processes including utilization of amine, sulfinol, iron sponge, and caustic solutions. Associated wastes may include spent amine, glycol 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 are exempted wastes and may be disposed of by injection back into the formation.

The disposal of excess drilling fluids and produced waters by evaporation, road spreading, and land farming may have an effect on the quality of surface waters. Runoff may allow the migration of chlorides and oily wastes into them.

Findings from Studies - With respect to the landspreading of liquid and solid wastes, two primary concerns are their salt content and hydrocarbon content. Studies by Freeman and Deuel (1986) and EPA (1987) have shown that soil/water mixtures with soluble salt levels below roughly 3,000 ppm total dissolved solids, exchangeable sodium percentage of less than 15, and a sodium adsorption ratio of less than 12 cause no harm to soil, vegetation, surface water, or groundwater. Landspreading of wastes resulting in oil and grease concentrations of up to one percent by weight in the waste/soil mixture are not harmful and will biodegrade readily. Repetitive disking and nutrient addition can achieve a soil mixture of these concentrations.

In stream monitoring by the Daniel Boone National Forest in Kentucky has found high concentrations of brine below oil production well fields. In Texas, heavy sediment deposites in streams were traced to gas well pads and service roads.

Reliability of Findings - Findings are reliable with respect to the existence of satisfactory legislation and regulations to lessen the risk of drinking quality groundwater and surface water contamination by oil and gas exploration, development, and production. There were no findings concerning the degree of enforcement of regulations.

Because little research was found, it is not possible to state the reliability of findings. However, there are numerous potential water quality impacts, water quality monitoring of oil and gas exploration and production operations is needed to detect impacts on domestic water supplies.

Ability to Extrapolate Findings - Because findings relate to national policy, they can be generalized across the Forest Service and other federal and state agencies that regulate oil and gas exploration, development, and producion on Forest System lands.

Secondary Linkages - Important linkages are road networks and related facilities that support oil and gas field operations.

Ability to Address Issues – The most significant minimum requirements that Class II wells must meet are: 1) only approved exploration and production wastes may be injected; 2) no well may endanger USDWs; 3) wells must be permitted before construction, unless otherwise permitted by rule; and 4) all wells must periodically demonstrate that they have no significant leak in the casing, tubing and packer and no significant fluid movement into an USDW through vertical channels adjacent to the injection wellbore.

A review of statutes, rules and regulations of all oil and gas producing states reveals that each provides regulatory agencies the right of access to inspect producing properties for regulatory compliance and to investigate complaints. If problems are identified, all states have the authority to: 1) issue cease and desist orders; 2) assess or seek administrative, civil or criminal penalties; 3) order cleanup activities; and 4) if necessary, ban further operations and sever an operator's pipeline connection.

State and Federal regulations normally require waste and reserve pit construction to comply with specified land use standards. They normally restrict pit usage to only the drilling operation and require that pits be closed within six to twelve months after cessation of drilling operations.

Minimum requirements for underground injection control programs can be found in 40 CFR 144-146. They identify specific construction, operation, and closure requirements for oil and gas production activities. These include requirements for casing and cementing, plugging and abandonment, and monitoring of injected fluids and the mechanical integrity of wells. Exploratory and development activities are approved by the surface managing agency with stipulations to assure proper site reclamation and rehabilitation. With the assistance of the Bureau of Land Management (BLM), EPA and the appropriate state regulators, precautions are taken to prevent contamination of acquifers by requiring casing of the hole penetrating the particular section. These agencies also assure satisfactory hole closure should the well become abandoned.

EPA's RCRA regulations include a "mixture rule" whose intent is to prevent the avoidance by oil and gas operators of hazardous waste regulations through dilution. This rule provides that commingling of any listed hazardous waste with a nonhazardous waste stream renders the entire stream a hazardous waste and subject to the provisions of Subtitle C of the Act.

In a 1988 Regulatory Determination, EPA identified regulatory gaps in State and Federal regulations for the oversight of certain waste treatment methods including land farming, road spreading, pit construction, surface water discharges, and abandonment practices. Consequently, it formulated a strategy for filling the gaps in State and Federal regulatory programs. It would improve Federal programs under existing authorities in Subtitle D of RCRA, the Clean Water Act, and the Safe Drinking Water Act. It would work with the States to encourage changes in their regulations and enforcement to improve some programs. Finally, it would work with Congress to develop any additional statutory authority that may be required.

Under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), EPA has broad enforcement authority to require Potentially Responsible Parties (PRPs) to perform cleanup of releases of hazardous substances or to recover from PRPs the costs incurred by the Agency to perform the remedial actions. Thus, CERCLA provides operators with a significant economic incentive to properly dispose of their solid wastes.

The National Environmental Policy Act requires stringent environmental review in either the form of an environmental assessment or an environmental impact statement, if warranted, for major federal actions permitted by agencies of the federal government when those actions may significantly affect the quality of the human environment.

Regulations promulgated by the Federal Oil and Gas royalty Management Act require oil and gas operators on federal lands to maintain site security and construct and operate wells and associated facilities in a manner which protects the environment and conserves the federal resource.

Findings suggest that State and Federal regulatory procedures would seem to lessen the risk of groundwater and surface water contamination with respect to oil and gas exploration, development, and production. Regulations, however, are effective only to the degree to which they are enforced.

Existing information appears satisfactory to enable managers and policy makers to effectively deal with the effects of oil and gas exploration, development, and production on drinking water quality.

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Chapter 10

Constructed Water Facilities and Infrastructure

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Introduction - This chapter addresses the effects of manmade activities such as application of sewage effluent, dewatering, dredging, and effects of structures such as dams, headgates, reservoirs, canals, water wells, splash dams and log flumes used to move logs through natural stream channels, upon the quality of "raw" drinking water before it arrives at the water treatment plant or settling basin. The term "hydromodification" is sometimes used to describe all of the kinds of activities which alter the natural flow of water. For pipelines, see chapter 11.

Issues and Risks - The Environmental Protection Agency ranked hydromodification as the third leading cause of water quality impairment to rivers. Only agriculture and municipal sewage treatment plants ranked higher (EPA, 1995, p. ES-12). Nationwide, there are over 68,000 medium and large dams built for hydropower, water supply, flood control and other purposes. The US Geological Survey estimates the cumulative storage capacity of these dams is almost 450 million acre-feet (Hirsch, pers. comm., 3/12/99). The Bureau of Reclamation manages about 600 dams and 53,000 miles of canals in 17 western states; the Army Corps of Engineers has about 700 dams and stores about one-third of all water in storage in this nation (Reetz, 1998, p. 127). The Forest Service has about 2,350 dams on its lands with a total storage capacity of about 55 million acre-feet. Half are owned and operated by the agency mostly for recreation, fire protection and fish or wildlife purposes. 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 purposes (Padgett, personal comm. 11/25/98).

There are also thousands of small dams in the United States that were designed and built to store domestic drinking water. Some of these reservoirs were built and are still operated for the single purpose of providing a reliable water supply. Since the 1940's, some existing and almost all new reservoirs became multi-purpose, to meet recreation, irrigation, flood control and sometimes hydropower needs, plus domestic drinking water. Often these other purposes create water quality problems for human health by altering water temperature, sediment transport, biological oxygen demand (BOD), chemical oxygen demand (COD), total dissolved solids, and streamflow.

The transport of water from one watershed to another can cause problems through channel erosion, sediment transport, and deposition in reservoirs and rivers. Subsequent dredging in large rivers and reservoirs often accelerate downcutting of headwater streams and de-stabilize stream banks, even where stream gradients are quite flat, such as Mississippi. Wetland is a form of hydromodification which can have water quality effects. Related information can be found in chapters 2 and 3.

Nearly all these hydromodification operations are influenced by water rights law which varies considerably from state to state. In most western states, water law requires users to obtain a state water right to diverted water out of streams or rivers. This removal results in higher water temperatures, lower oxygen levels, lessened sediment transport capacity and other water quality problems described by Getches, MacDonnell and Rice (1991). The riparian water rights doctrine used in states east of the 100th parallel does not generate water quality problems because most of the water is usually left in the stream. The subject of water rights law is beyond the scope of this paper and will not be treated further.

Findings – Hydromodifications can impact water quality with algae blooms, trihalomethane production, sediment transport and depostion, and in chemical and physical properties.

Dams and Algae Blooms

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, it quiets down , slowing the diffusion of oxygen from the air into the surface water and also slowing bacterial decomposition of organic matter within the water column. In turn, the biological oxygen demand (BOD) and chemical oxygen demand (COD) may cause anoxic (lack of oxygen) conditions to devlop in the water, especially in deeper water layers. This phenomenon has been well studied and detailed models which quantify this effect have been developed. Anoxic conditions generally cause secondary problems in drinking water, usually taste, smell, color that do not pose health risks, but which do increase water treatment costs.

The operation of dams can affect the likelihood of blue-green algae blooms which in turn can produce toxins that have been reported fatal to livestock, wildlife and pets. When dam owners begin to fill 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 water, conditions favor algae blooms. Toxic blooms of blue-green algae have been reported for many states, including Montana, both Dakotas, Indiana, Iowa, Minnesota, Missouri, and Wisconsin, even in forested and largely pristine watersheds (Horpestad et.al., 1977; Carmichael, 1981; Fawks et.al., 1994). Whether a given bloom will turn toxic is still unknown. Accidental ingestion by humans who are engaging in water sports such as waterskiing, jetskiing, diving, swimming is possible, although no deaths have been reported; prudence calls for prohibiting all water contact sports when blue-green algae blooms are present at public use areas. More powerful methods of detection have shown that the occurrence of toxic blue-green algae blooms is much more widespread than what has been believed based solely on reported poisonings.

Trihalomethane

Researchers (Martin et. al, 1993; Arruda and Fromm, 1989) report that reservoir and lake organic sediments can contain and release trihalomethanes (THMs). Trihalomethanes form when chlorine that is added to drinking water during the disinfection process reacts with certain naturally occuring organic molecules (THM precursors). THMs may cause cancer and genetic mutations in humans. Understanding THM precursor sources is important because limiting them may lower risks to human health and lower water treatment costs. Reservoir management may reduce chemical and operation costs and the need to chemically control algae, which has both economic and environmental advantages (Kortmann, 1989) . In one study in Ohio, all sediment samples had significant THM precursor releases relative to controls. Anaerobic conditions and deep water sediments had much fewer THM precursors than aerobic sediments from shallower zones. Karimi and Singer., 1991. and Wardlaw et.al. 1991 reviewed the role of algae as THM precursors. They found that a variety of organic compounds found in natural waters, especially humic and fulvic acids derived from soils and decomposition of plant material, are sources of THM precursors. No discernible trends in the ability of particular algae species to generate THMs can be drawn from published data. THM concentrations arising from a natural algal bloom could theoretically exceed maximum allowable concentration levels for drinking water.

Sediments Deposited in Reservoirs

The sediment deposited in reservoirs can also pose public health problems if it contains heavy metals, radioactive elements, or pesticides. Many of these are chemically bonded to sediment particles under the right set of conditions, such as low pH. Often the risk to human health remains low as long as the sediment remains undisturbed at the bottom of a lake or reservoir used as a source of drinking water. The accidental failure or deliberate removal of a dam may pose a human health water quality problem by the destabilization of the accumulated sediment, but literature is lacking on this topic. Changing streamflows, e.g., from changes in operating schedules at hydroelectric dams, may also have the potential to mobilize or deposit sediment which may then cause drinking water quality problems. In my opinion, further research needs to be conducted.

Splash dams and log flumes were constructed on many rivers in New England, the Lake States, and in the West. The dams were earthern structures less than 20 feet high with the main spillway constructed of wooden boards, and often held from a few hundred to 1,000 acre-feet of water. When the boards were removed, an artificial flood was released down the river, sweeping logs down the channel. Empirical evidence indicates the effects upon drinking water quality today are likely to be minor since so many years have passed and channels have recovered for the most part. The flumes were constructed of boards and were used to transport logs, ties, and mine props. Field evidence is they have no or very little effects on today's drinking water quality in the watersheds where they once operated.

Controlled removal of sediment by dredging of channels or from existing lakes or reservoirs can stir up sediments that can be a problem for domestic water suppliers. These sediments pose special problems if they contain toxic substances or if they are massively released such as after a dam failure or planned dredging.

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, an aqueduct or pipeline to be carried to some place of use, often miles away. The removal of this water results in changes in the remaining river water, such as increased concentration of pollutants, higher water temperatures, and subsequent increased biological activity of aquatic organisms. The pH of the water often rises as well, which can change the solubility of metals and rates of chemical reactions in the water column. Suspended sediment transport declines as flow declines, casuing increased deposition of fines in sand and gravel bed 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 to be treated at all! There is not much scientific literature on this particular aspect. The same is true for water added to stream channels by diversions from other watersheds or other aquifers. A complicating factor is differences in chemical, physical or microbiological water quality between the rivers when their water mixes together. The effects probably depend upon particular local conditions so it is hard to generalize. Much of the information cited in the rest of this report may be useful, but will need to be applied on a case-by-case basis.

Water Well Effects on Drinking Water Quality

High pumping rates from water wells can decrease flows of nearby streams that are water sources, thereby affecting the particle size and amounts of sediment in transport, along with whatever metals, pesticides and organic acids are sticking to the particles, pH, dissolved oxygen and water temperature. These wells can drawdown the ground water table which can affect the yield and water quality at other wells tapping the same ground water or aquifer.

Wells located on floodplains can become contaminated during floods if they are not properly protected ahead of time. Singer et.al, 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. Also, 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, cement and bentonite clay, have properties that can be aggravated by the drilling and well construction methods in certain hydrogeologic environments. There is a large body of scientific literature on well construction and maintenance, as well as that industry's own materials, that the reader should consult if it appears that local water wells are responsible for, or help transmit, pollutants in forested and rangeland watersheds.

Sewage Effluent and Sludge Applications to Forest and Rangelands

A fair amount of research on this topic was conducted in the 1980's in the Lake States and southeastern U.S. often under hardwood forests. An excellent reference is The Forest Alternative for Treatment and Utilization of Municipal and Industrial Wastes, edited by Dale W. Cole, Charles L. Henry, and Wade L. Nutter, University of Washington Press, Seattle, 1986. Studies by Brockway, 1988; Machno, 1989; Sorber and Moore, 1986; Brockway and Urie, 1983 dealt with effects of applying municipal, including papermill, sludge and/or wastewater to forests and monitoring the movement of nitrogen and other variables in the leachate and ground water. Results included nitrate nitrogen concentrations exceeded the 10 mg/ liter potable water standard in ground water under aspen plots treated once with 16 or more Mg/ha undigested papermill sludge, and under pine plantations receiving 19.3 Mg/ha anaerobically digested municipal sludge in a single application. Brockway and Urie (1983) estimated safe applications of anaerobically digested municipal sludge could be applied to a red pine and white pine plantation at 16.5 dry Mg/ha (990 kg total nitrogen per hectare) or less, and to aspen stands at rates up to 19 dry Mg/ha (1140 kg total nitrogen per hectare) that would keep ground water nitrate nitrogen levels under the 10 mg/liter nitrate standard. Units for converting these data are: 1 kg/ha = 0.89 pounds per acre; and 1 Mg/ha = 0.44 tons per acre or 890 pounds per acre.

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

Harris-Pierce et.al. 1995 applied sewage sludge on a semi-arid grassland in Colorado and found that increasing rates of single applications from 0 to 22 to 41 Mg/ha 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, but that all constituents remained below EPA's drinking water and livestock watering standards. In their paper, Burkhardt et. al. 1993 argue for a careful approach to the use of range lands for applying sewage sludge (bio-solids) because of the little opportunity for nutrient uptake and bio-solids assimilation by the native vegetation and low rainfall amounts. Irrigation could overcome these limitations in their opinion.

Spray applications of treated municipal wastewater on forested lands in Michigan has been studied by Urie et.al 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. Whether pathogenic organisms can move through forest soils to ground water and remain viable for long is not as well studied and reported upon. Likewise, effects of draining wetlands on drinking water quality has not been researched. Logic would say that the drainage water can be made potable at the treatment plant.

Reclaimed Water and Return Flows

After being used, water diverted from surface or ground water is often returned to these sources. Water 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 also regulations on sewage treatment because it is a point source of pollution under the Clean Water Act, as amended. The re-use of water effluent from sewage treatment plants is growing in the United States and has passed the one billion gallon per day level for both nonpotable and potable purposes. It is included in this chapter since it logically links to the water facility infrastructure within watersheds that contain towns and cities. Water reuse for nonpotable applications, such as irrigation, lawn watering, car washing, toilet flushing, is widely accepted where water supplies are scarce, especially Arizona, California, Florida, and Texas. Recommended limits have been set for many physical parameters and chemical constituents of nonpotable water by the EPA and National Academy of Sciences following many studies. The health risks from disease-causing microorganisms are not as well known; hence, there is no direct potable reuse practiced anywhere in the U.S. (Crook, 1997)

Indirect potable reuse occurs via surface water augmentation when effluent that has received advanced treatments (chemical clarification and two stage recarbonation with intermediate settling, multi-media filtration; activated carbon adsorption; ion exchange for nitrogen removal; and breakpoint chlorination) is returned to a receiving river or lake. It then mixes with natural raw water and later removed as drinking water downstream. Indirect potable reuse can also occur via ground water recharge (injection wells); however some states prohibit that if potable aquifers would be contaminated. Other states have set stringent water quality limits and require high levels of effluent treatment before dischage to the aquifer is allowed to occur (Crook, 1997).

Crook (1997) also lists a number of references in his article that would be very helpful to the reader who needs to deal with water reuse in their source watershed.

Irrigation return flows can be a major human health risk if they contain pesticides, heavy metals or other toxic substances they acquire from atmospheric deposition and the irrigated soils and plants. There is a very large amount of literature on this subject, which is outside the scope of this report.

Reliability of Findings - The scientific literature on direct effects of dams, water diversion and conveyance structures, water wells, and other related man-made structures upon human drinking water quality is very limited. There is far more known about their effects on physical habitats of aquatic life forms and the biological responses. However, the indirect effects of the water infrastructure are known, have been much more researched and are included in this chapter. The reader can have confidence when trying to apply these findings to their local situations in a reasonable manner; of course there will always be exceptions and other factors present which can lead to a different answer.

Most of the studies included seem to lack a thorough description of how the water facility was operated and whether the manner of operation could have, or did, influence the results of the experiments or observations. It seems desirable that facility operational details should be better evaluated in future research studies.

Ability to Extrapolate Findings - The indirect effects of dams, water diversions and conveyance structures, water wells should apply in all forested and range land watersheds in the United States. Some of the magnitudes and timing of the indirect effects will vary by region and perhaps by altitude due to temperature and precipitation regimes that control rates of many chemical reactions and biological processes, such as algae bloom frequency and extent. 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 in all cases.

Secondary Linkages - The existence, operation, and maintenance of the constructed water infrastructure of the United States discussed in this chapter does have multiple and sometimes subtle linkages to the quality of drinking water and human health. Some are well known and reported upon; other ties are apparently rarely or never researched and reported in the scientific literature which I perused. It may be the cumulative effect of these structures or facilities which ultimately cause undesirable impacts on drinking water quality and/ or quantity.

Ability to Address Issue - Owners of dams that 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., even in very small amounts. Most of the other risks to human health from drinking water quality management at all manmade water storage and control structures are known and can be assessed in local watersheds by applying information provided in this paper by professionals in hydrology and health sciences.

Management of risk to drinking water supplies from applying treated sewage sludge on forested and range lands is very possible as long as the information referred in this paper is applied in a reasonable manner. The important thing is to insure that disease causing organisms have been killed at the sewage treatment plant before the sludge is hauled to the forest or range and applied to the soil.

Since there are often so many water wells in the rural areas of interest in this paper, the ability to manage them so they do not become sources of contamination to ground water themselves, is difficult. People doing source water assessments in forested and range land watersheds should not neglect these wells during the vulnerability phase of the assessments.

Research Needs - The direct effects of dams, water diversion and conveyance structures, and water wells upon human drinking water quality needs to be researched, including their manner of operation. Why some blue-green algae blooms turn toxic and other similar looking ones do not, needs to be investigated.

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Chapter 11
Roads and other corridors

W. J. Elliot

The focus of this chapter is on the impact of roads and other corridors, including trails, utility rights of way, railroads and airfields in forest and grassland watersheds on water quality.

Introduction - Roads represent a major feature in most watersheds. Roads are essential for a wide range of access including residential, recreational, and managerial. Roads are needed to access oil fields, mines, and forests (USDA Forest Service 1998). They can be public, managed by local, state, or federal agencies, or private, managed by individuals or industries. In addition to roads, similar watershed disturbances associated with construction, maintenance, and management also occur from forest harvesting and fire (Chapter 4), parking lots and large roofed areas (Chapter 6), mineral development (Chapter 8), airfields, and utility rights of way.

Figure 1 is a diagram of a typical insloping road, common in steeper terrains. Water from the traveled way 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 water. The collected water is then delivered to a flat or concave hillslope to infiltrate, or to a natural channel which is part of the stream system (Packer and Christensen 1977).

There are a number of common water quantity and quality issues associated with roads and similar corridors. Generally, the disturbed and compacted road traveled way has much greater runoff rates and amounts from precipitation and snowmelt than less undisturbed parts of the watershed (Elliot and Hall 1997). The traveled way and the ditch are the main sources of detached sediment (fig. 1)(Tysdal and others. 1997) from surface erosion, whereas the cutslopes and fillslopes are major sources due to mass wastage.

There is usually limited surface runoff from undisturbed forests on the uphill side of a road, and there is generally considerable infiltration and sediment deposition on the original surface below the road (Morfin and others. 1996; McNulty and others. 1995; Packer and Christensen 1977; Tysdal and others. 1997). This principle of buffering between a disturbance and a stream applies to most disturbances, although expecting buffering to solve all sediment risks from watershed disturbance is unwise. Areas of greatest pollution risk to water supplies are stream crossing by roads, railroads, or pipelines, and any other activities that are close to streams. In the context of groundwater, water supplies from shallow wells, or in highly porous geologies, such as karst (Gilson and others. 1994; Hubbard, Jr. and Balfour 1993; Keith 1996), are the most susceptible to pollution from road-related activities.

The excavation at the bottom of the cutslope can intercept groundwater which can lead to instability of the road or the cutslope, and impact hydrology as discussed in this chapter (Burroughs, Jr. and Marsden 1972; Jones and others. 1999). The interception of groundwater at the base of a road cut is also a potential source of pollution from acid drainage as acid-generating minerals are exposed to and rainfall (Chapter 9). All of the excavated surfaces revegetate slowly and are prone to erosion (Burroughs, Jr. and King 1989; Grace, III and others. 1996).

Erosion and hydrologic events associated with the impacts of disturbances on the landscape are dependent on local climates, geologies, topographies, and management practices. Even though the dominant processes do not change, the magnitudes of factors driving those processes do change.

Runoff and seepage from roads and rights of way can contain elevated levels of sediment, metals, complex hydrocarbons from the highway material and traffic, and traces of pesticides. This chapter discusses the impacts of roads and other corridors on runoff, surface, and subsurface water quality and quantity.

Altered Hydrology

Issue - The presence of a road network in a watershed may increase the frequency and magnitude of peak discharge events, particularly on small watersheds. There is also concern that road networks increase total runoff and decrease the time to peak runoff from major events.

Findings - Roads have a number of impacts on hydrology. They intercept precipitation and snow melt, and because they have lower infiltration rates, divert it as surface runoff to channels (Packer and Christensen 1977). Cutslopes (Fig. 11.1) can also intercept groundwater from subsurface flow paths to surface runoff. Burroughs and others. (1972) calculated that 58 percent of the runoff from a road during the spring snowmelt season was due to intercepted subsurface flow in a high-elevation study in the northern Rockies. Megahan (1972) found that road segments on granitic soils in Central Idaho collected about 8.4 inches of water in subsurface flow from the area above the road. Road ditches can also extend the stream network, increasing the volume of water available during the early part of a runoff event, and the presence of roads may shorten the time to peak flow during a runoff event (Wemple, Jones, and Grant 1996).

If a road culvert is too small, or becomes blocked, water can be diverted from one subwatershed to another, and in the process may cause severe ditch and channel erosion (Furniss, Love, and Flanagan 1997; Megahan 1972; USDA Forest Service 1998). The cumulative effect of these increased runoff, extended stream networks, and channel diversions is believed by some scientists to increase in frequency and magnitude of peak discharge events by the extension of the stream network by road ditches (Megahan 1972; Jones and Grant 1996; USDA Forest Service 1998):

Adding gravel to the road surface, which decreases runoff rates (Elliot and Hall 1997; Foltz 1996), can reduce these hydrologic impacts. Also, ripping closed roads can increase road infiltration rates, although the rates were not increased to undisturbed levels (Luce 1997). Designing roads with culverts or surface drainage delivering water to hillsides rather than to channels will also reduce these hydrologic impacts (Tysdal and others. 1997).

Reliability - There was only one study on which much of the watershed information is based (Jones and Grant 1996; Thomas and Megahan 1998). This was a small paired-watershed study in the Oregon Cascades (Ecoregion M242B). There is ongoing controversy about the appropriateness of some of the experimental design from which these data have been collected (Jones and others. 1999; Thomas and Megahan 1998). Studies of this nature generally require small paired watersheds, and the natural variation and past history of watersheds may be sufficient to mask differences due the presence of roads (Jones and others. 1999). The greatest challenge in applying the hydrologic findings is that landscapes are highly variable. Even though information from a study may support some of these findings, differences in hydrology due to the presence of roads may not be easy to isolate.

The studies associated with gravel and ripping impacts, however, can be applied more widely. Numerous studies have measured reduced erosion with the addition of gravel (Swift, Jr. 1984b; Yoho 1980), and it is likely that much of this reduction is associated with reduced runoff (Foltz 1996)

Ability to Extrapolate Findings - Generally, roads will have the same impact on hydrology regardless of climatic or soil differences. The disturbances caused by road construction, including removal of surface organic layers and topsoil, followed by compaction, apply to all roads. Impacts of disturbances are likely to be greater in wetter climates, as shown in table 11.1.

Secondary Linkages - Increased road runoff generally leads to increased road surface erosion, increased erosion on hillsides and channels below roads, and increased sediment delivery to streams (Tysdal and others. 1997). Hydrology dominates the sediment and pollutant delivery processes, so any activity, like a road or similar disturbance that increases runoff will increase the likelihood of delivery of pollutants to a stream system.

Another linkage associated with roads or other excavations on steep terrain is that groundwater flow may be reduced because of interception (Burroughs, Jr. and Marsden 1972). This is only likely to be a problem in conditions where shallow aquifers or springs are a source of water at the base of watersheds with dense road networks.

Railroads will be similar in their impacts, except that the contributing surface area is reduced. Other disturbances, such as airfields or other inholding activities that have compacted surfaces will have similar effects on the hydrology (Chapter 8).

Ability to Address Issues - Research is ongoing in this area at scales ranging from plot to landscape, and new tools for determining road impacts on hydrology are under development ranging from site-specific to landscape scale models. Before any models are used to address issues, they should be validated for a watershed similar to the one in question. In the past, the impacts of roads on watershed hydrology have not been included in hydrologic planning. Future designs of hydrologic structures may need to be more conservative if watersheds are highly developed.

Research Needs - The main need for research is to develop watershed scale studies to compare relatively undeveloped watersheds to similar watersheds with greater disturbances. Such sites are difficult to find, so efforts to develop predictive models that allow comparisons of disturbed and undisturbed watershed hydrology need to be developed and verified.

Sedimentation

Issues and Risks - In most watersheds, sediment is the greatest pollutant. Roads are a major source of that sediment in most forested watersheds (Appelboom and others. 1998; Megahan and Kidd 1972a; Megahan and Kidd 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 associated with watershed disturbances into the surface and ground water systems (Boxall and Maltby 1997; Gilson and others. 1994; Patric 1976; Thomson and others. 1997; Wuest, Kern, and Herrmann 1994).

Sediment may be from surface erosion, which is generally more likely to carry pollutants. In steeper watersheds, more sediment may be from mass wasting, which tends to result is greater volumes of soil entering the stream system, causing downstream problems with sedimentation.

Findings - Numerous researchers and managers throughout the U.S. have identified roads as a major source of sediment in watersheds that are generally undisturbed, like forests or range lands (Table 11.1). Soil erosion is a complex process (Figure 11.2) resulting from the interaction of climate, soil, topography, and vegetation, with disturbances to the system leading to major erosion events. Tables 11.1 and 2 present some typical erosion rates and sediment plume lengths for different regions in the U.S. for different types of disturbance. Note that some investigators have reported erosion rates specifically for roads, ranging from 5 to 550 t/acre, whereas others have reported erosion rates of watersheds containing roads in the range of 0.02 to 2 tons/acre. Roads generally account for 1 to 5 percent of a watershed surface, accounting for some of the difference between erosion rates. The remainder of the differences likely stem from deposition of eroded sediment on hillsides and in channels.

Table 11.3 presents an example study of the relative impact of roads on the sediment budget of a mixed rural and urban watershed in northern Idaho (ID DEQ 1997). Even though a road may only be 1 percent of a watershed, even in an intensively managed, watershed, it can still be a source of 8 percent of the sediment. Megahan (1974) estimated that in central Idaho, the sediment yield from watersheds without roads was about 0.07 t/sq mi. per day, whereas the presence of roads increased this yield by a factor of five. McNulty and others. (1995) attributed the majority of sediment from a forested watershed to unpaved forest roads in a southeastern watershed.

Immediately following construction, erosion rates are high due to bare slopes and road surfaces. 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, Jr. and King 1989; Grace, III and others. 1996; Ketcheson and Megahan 1996). Road surfaces, however, will likely continue to be a source of sediment as long as there is traffic or maintenance that prevents the establishment of vegetation (Elliot and others. 1996; Swift, Jr. 1984a; 1988). Graveling can decrease erosion rates by up to 80 percent (Burroughs, Jr. and King 1989; Swift, Jr. 1984b), 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 road traveled way, and the fill slope as sources of sedimentation (Figure 11.1). They suggested mitigation measures for each of these components, including mulching, geotextiles, seeding, and sodding. There are many other studies which demonstrate the effectiveness of these treatments (some typical studies presented in table 11.4), and their application are included in many state-recommended practices.

Wemple and others. (1996) and Tysdal (1999) expanded the area affected by road erosion from the immediate road prisms of most previous studies, and demonstrated that the hillside or channel below the road could be an area of deposition, or could be the major source of sediment from a given segment of road. The excess runoff from roads can overload stable ephemeral channels resulting in significant downcutting of the channel (Sidebar 1).

In areas of high rainfall, like the Coastal range in Washington and Oregon (ecoregion M242A), more sediment comes from roads due to landslides than from surface. Beschta (1978) reported that sediment yields increased from around 300 t/sq. mi. per year before roads and harvesting, to about 400 t/sq mi per year after installing roads and harvesting. Much of the sediment in this high-rainfall area was from mass failure. In a recent study in the Clearwater National Forest in Idaho (ecoregion M333D), for example, 58 percent of the landslides that occurred were associated with roads (McClelland and others. 1998). While surface erosion is an ongoing source of sediment, land slides tend to be epicotic in nature, contributing 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. (1997) calculated 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, like pipelines can also be potential sources of sediment (Gray and Garcia-Lopez 1994; Sonett 1999),. Any construction that exposes bare mineral soils, particularly on sites that are adjacent to ditches or streams, is likely to increase sedimentation to values near the maximum observed for new roads in Table 11.1. The values predicted in Table 11.1 can be considered as a maximum, as exposed subsoils will generally have lower erosion rates, and topsoil erosion rates will approach that of roads if compacted by construction traffic. Once installed, rights of way may continue to be sources of sediment if vegetation or other erosion control practices are not initiated (Burroughs, Jr. and King 1989; Gray and Garcia-Lopez 1994) (Table 11.4). Frequently off-road vehicle enthusiasts may use rights of way for recreation. The resulting erosion rates from the compacted trails will be similar to those shown for roads in Tables 11.1 and 11.2.

Other established trails such as bicycle, foot, or horse trails can also experience eroding at rates similar to roads (Leung and Marion 1996; Webb 1996). The total sediment delivered from these trails will generally be lower, however, because the total surface area of the narrow trails is less than that of most roads.

Much of the sediment eroded from a road never reaches a stream system, but is rapidly deposited near the road. Rummer at al. (1997) found no significant sedimentation effects beyond the clearing limit of the road in a bottom land hardwood study on a flood plain (ecoregion 231A). Brake and others. (1997) found that average sediment plume lengths below forest roads in Oregon (ecoregion M242A) were 16.7 ft for old roads, 30.5 ft for new roads. Table 11.2 shows the distance associated with sediment travel below roads in several studies. Numerous scientists have developed equations from field observations to predict how far sediment will travel (Ketcheson and Megahan 1996; Packer and Christensen 1977; McNulty and others. 1995; Swift, Jr. 1986). In most of these studies, the authors observed considerable variation in length, and the predictive equation that was developed is limited to the local conditions, and should not be applied to other areas with different climates, soils, and topographies.

Various mitigation measures to reduce road erosion are commonly prescribed by state and federal agencies. The most common methods include surfacing the road with gravel, decreasing the spacing of cross drainage, locating roads further farther from streams, or limiting road gradients (Burroughs, Jr. and King 1989; Seyedbagheri 1996; Swift, Jr. 1984b). Treatment of cutslopes and fill slopes has also been effective in reducing sediment delivery from new roads (Table 11.4)(Burroughs, Jr. and King 1989; Grace, III and others. 1996). 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).

There have been a number of prediction tools developed to estimate the amount of sediment that leaves forest roads. A number of site-specific models were developed in the Northern Rocky Mountains (ecoregions M331, 332, and 333) by Forest Service hydrologists, and the most recent is the WATSED model (USDA Forest Service 1990). McNulty and others. (1995) presented a GIS-based methodology for predicting sediment delivery from a road network, and observed that additional work with physically-based models is necessary to improve the prediction of sediment delivery from roads (ecoregion M221D).

The Water Erosion Prediction Project (WEPP) model is a physically based model 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 to a wide range of conditions as long as 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. (1998; 1999) have released a simplified tool aid managers in determining the impacts of climate, soil, and topography on the delivery of sediment from roads, which included 33 different climates. An easy to use interface for the WEPP model for forest roads is under development, and is now available on the WWW to aid in determining sediment delivery from forest roads (Elliot, Hall, and Scheele 1999). Predicted road erosion rates and sediment plume lengths from this web site are shown in table 2 for comparison with observed erosion rates.

Reliability - Researchers worldwide have measured increased sedimentation from roads and similar disturbances, and the role of disturbance in sedimentation generally accepted as commonplace. Magnitude of erosion may vary considerably with climate, but the relative impacts of soil, topography, and management are generally the same (Elliot, Foltz, and Luce 1999; Morfin and others. 1996). Observed erosion rates are highly variable (Tables 11.1, 11.2 and 11.4) due to the high natural variability in the factors that cause erosion (Figure 11.2). Even a well-designed experiment frequently results in variations from the mean of up to 50 percent (for example Elliot and others. 1989). This high variability should be considered when interpreting any research or monitoring results, or any erosion prediction value. The general level of confidence of any observed value is that it is within 50 percent of the mean, hence the accuracy of any predicted value is likely to be within plus or minus 50 percent (or more) of what may be observed. The example WEPP-predicted values presented in table 11.2 are generally within this range.

Managers should exercise caution when applying any model to an area where it has not received some validation. When interpreting any validation, however, the large range in observations due to natural variation must be considered for both the observed values and the predicted values.

Ability to Extrapolate Findings - The increase in sedimentation from disturbances has been widely measured, and the presence of a road or similar disturbance is likely to be a source of sediment to surface waters under all but the most arid conditions. The management principals and effects of mitigation practices will be similar, but the actual amounts of sediment detached and delivered, will vary widely (tables 11.1 and 11.2). Although most studies have focussed on roads, the impacts of disturbance also apply to other watershed activities. All construction activities, and any maintenance that denudes hillsides of vegetation will increase erosion. The same mitigation measures as common for roads can be adapted for any watershed disturbance.

Secondary Linkages - Eroded sediments frequently carry attached pesticides and other hydrocarbons from roads, airfields, or other disturbed areas. Sediment concentrations may serve as a simple surrogate measurement for more complex pollutants in road runoff (Thomson and others. 1997). Sediments may accumulate in sediment basins, stream corridors, or reservoirs, releasing the chemicals at some future date. Airborne dust from roads may also contribute to offsite transport of chemicals if the surface is contaminated (Christensen and others. 1997).

Sediment from roads entering streams can trigger channel instability. In a recent follow-up to a landslide event (Elliot and others. 1994), it was found that much of the stream channel was aggrading where sediment had been deposited in the stream system. In other studies, deposited sediment may be a source of increased sediment yields from the watershed for many years beyond the disturbance (McClelland and others. 1997) as channel side slopes collapse, or headcuts and scour remove sediment from channel bottoms.

Seasonal closure of roads is frequently practiced to minimize damage to roads during wet seasons. Although such a practice will likely result in lower sediment yields, it may adversely impact other activities within the watershed, such as residences and recreation (Chapter 6).

In many forested watersheds, road removal is common. Generally this practice will reduce the risk of sedimentation, but there are risks associated with surface erosion and mass wasting if removal practices are not properly planned or carried out (Elliot and others. 1996).

Ability to Address Issues - Sedimentation and its prevention and treatment have been widely researched. Generally, there are methods to reduce sedimentation to acceptable levels. Most states have developed recommended management methods to minimize sedimentation both from roads, and in many cases, from related construction sites (Aust 1994; Seyedbagheri 1996). Unfortunately, some of these methods may be expensive. Watershed and water supply managers need to balance the role of upland prevention and onsite treatment to determine the best combination of practices to ensure quality water.

Research Needs - Upland erosion and sedimentation processes are generally well understood. The impacts of sediment attenuation on the hillsides between sites of disturbance and streams, and sedimentation within and through stream networks are not well understood. Future research focusing on sediment transport and storage in stream networks will enhance our ability to address sedimentation issues in watersheds.

Pesticides

Issues and Risks - Surface runoff and groundwater seepage from rights of way may contain pesticides from weed control or other management activities.

Findings - Pesticides may be applied to road or railroad banks or other rights of way. If best management practices are followed, there is usually little risk of any offsite pollution (Chapter 5). Such practices include not spraying in the vicinity of exposed water, and not spraying during rainy weather. Eroded sediments carry most pesticides that enter the stream system from other than direct contact. Practices to minimize erosion on rights of way will also minimize the risk of pesticide contamination. Ellis and others. (1997) found that pesticides were only pollutants that could impact groundwater as well as surface waters in a United Kingdom study.

In some areas with shallow groundwater sources, the presence of road ditches may increase the risk of groundwater pollution. Road ditches have a shallower depth of soil to buffer the groundwater from pollution by herbicides used in controlling weeds in ditches. Kohl and others. (1994) measured two to five times the amount of 2,4-D and picloram passing through a soil column typical of a road ditch, compared to a typical agricultural soil column. Also, the amount of clay and organic matter in ditches may be lower than in most soils, again reducing the ability of the bottom of the ditch to attenuate or prevent the passage of chemicals into the groundwater. Because of the low conductivity of roads and road sideslopes, there is an increased risk of runoff carrying pesticides into surface water systems (Heather and Carter 1996)

Reliability - Most studies have focused on specific sites. Measuring chemical concentration rates are difficult and expensive. In some cases, it may not be possible to separate background rates from pollution due to the presence of a disturbance. Measurable pesticide concentrations may adversely impact local aquatic ecosystems, but may not be in high enough levels to cause concern for public water supplies (Chapter 2).

Ability to Address Issues - The processes that impact chemical movement are a complex interaction of climate, soil, vegetation, topography, management activities and chemical properties. This makes it difficult to translate the results of one study to another site.

Secondary Linkages - The risk of pesticide pollution is linked to management practices around a disturbance. Pesticide management practices are more fully discussed in Chapter 5.

Forest roads have been identified as corridors for infestation of exotic weeds (USDA Forest Service 1998). Control of infestations may require increased use of selected herbicides along roads, increasing the risk of offsite pollution.

Ability to Address Issues - Pesticide applications practices are well known, and most states have in place guidelines to minimize off site impacts of pesticide application. Following label instructions is vital to ensure minimal risks to the operator and the environment from the pesticide. Skilled operators who have been trained in proper application procedures generally apply most pesticides along roads and other corridors.

Research Needs - See chapter 5.

Hydrocarbons, Cations, and Related Pollutants

Issues and Risks - Runoff from roads and similar surfaced sites can contain a host of hydrocarbon and other chemical pollutants, either adsorbed to sediments (Boxall and Maltby 1997; Sansalone and Buchberger 1995), as particles, or absorbed by the runoff (Ellis, Revitt, and Llewellyn 1997). These chemicals can find their way into surface and subsurface water supplies.

Findings - Researchers in many areas in the world have identified a range of chemicals in road runoff (table 5). An example of the concentrations of a number of the pollutants in presented in table 6 from a study with a database from Minnesota (ecoregions 212, 222 and 251). Some of the pollutants are from the road material itself, some occur in the soils and rocks on the site and are released during the construction or subsequent erosion processes, and many are from vehicles. Traffic and road surfacing may contribute undesirable cations, hydrocarbons and metals to surface and subsurface waters (Baekken 1994; Boxall and Maltby 1995; Maltby and others. 1995; Mungur and others. 1994). Most studies on the impact of roads and similar disturbances have focused on sites that were heavily trafficked like major freeways (Mungur and others. 1994). If water source areas contain major roads, then runoff treatment may be necessary to ensure that undesirable hydrocarbons do not enter the water supply source.

In areas with acid rain (See Chapter 3), one of the impacts of road runoff containing salts is that mercury in the soil can be mobilized and transported to groundwater (MacLeod, Borcsik, and Haffe 1996). Similar mobilization can occur from fertilizer salts common in agriculture. This risk will be limited to areas that have elevated concentrations of mercury in the soil column. A beneficial impact of the presence of roads in acid rain areas is that cations can be released from the road that have a buffering effect on the runoff acidity. Morrison and others. (1995) measured pH values in road runoff water from small events ranging from 6.0 to 7.0, compared to the average rainfall pH of 4.1.

Ions from deicing or dust control chemicals are common pollutants from the road surface (Church and Friesz 1993; Kjensmo 1997; O'Malley 1998; Pugh, IV and others. 1996). Road salt contamination has become a serious problem, particularly in the northeast and Midwest U.S. (Church and Friesz 1993).

Church and Friesz (1993) state that in Massachusetts alone, there were complaints from 100 of the 341 municipalities in the state in a seven year period, and that nationally, about $10m is spent each year to prevent or remediate problems associated with road salt contamination. Surface waters are not as vulnerable as groundwater to such contamination as there tends to be much greater dilution and mixing in turbulent channels carrying runoff from roads (Jongedyk and Bank 1999). Calcium Magnesium Acetate (CMA) and Potassium Acetate (Kac) are deicing chemicals that are most benign in the environment because they contain weak biodegradable acids. Sodium Chloride, Calcium Chloride, and Magnesium Chloride, however, leave residues of chloride ions on the highway surface that may contaminate surrounding groundwater (Jongedyk and Bank 1999).

Some of these ions (Ca, Mg, and K) can enhance vegetation growth along highways (Pugh, IV and others. 1996). Although many thousands of tons of salt are spread annually on highways, concentrations of salt in runoff are not likely to be a major source of pollution for drinking water, even though impacts on the aquatic ecosystem may be great. The proximity of the road to surface water has a major affect of the likelihood that ions will reach surface water. In some cases, elevated levels of cations from deicing in the road runoff, like sodium, may be adsorbed by the soils near the road, and pose no further concern to the aquatic ecosystem (Shanley 1994). In Norway, Kjensmo's (1997) study found increased salt concentrations in a lake adjacent to the road but not elsewhere. Pugh and others. (1996) observed that ion concentrations declined exponentially away from an interstate highway intersecting a peat bog.

Measuring concentrations of a number of pollutants can become tedious and expensive. To reduce the cost, surrogate relationships between more easily measured pollutants, such as suspended solids (mainly sediment) or dissolved solids and specific pollutants have been developed (Thomson and others. 1997). Such surrogates may be useful if relationships were developed for nearby conditions, but become less reliable when interpolating to other regions.

O'Malley 1998 reported that oil-based dust suppressants may be environmentally more risky than salt-based products. A literature search for the Forest Service (Heffner 1997) on water quality effects of three dust abatement compounds reported that calcium and magnesium chloride showed some toxicity towards 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 use these compounds, the use of one over another would be more dependent on cost, availability, and local conditions than effects to the environment". Instructions for application risks to health of dust control and other stabilization chemicals should be carefully read to minimize any offsite impacts on water quality.

Groundwater as well as surface water can be polluted by contaminated runoff from disturbed sites. Levine and others. (1997) found both jet fuel and other hydrocarbon pollutants in groundwater near a Florida Air Force Base. A common practice in some areas is to dispose of surface runoff in large soakaways (Price 1994). Although this practice eliminates concerns about surface water hydrology and quality, it increases the risk of groundwater pollution. Price (1994) observed that even though there was a potential for polluting a public water supply through such practices, that none had ever been reported in the United Kingdom. Ellis and others. (1997) found that the only pollutants associated with roads and similar activities that impacted groundwater were pesticides.

Gilson and others. (1994) completed research on the effectiveness of filter systems for highway runoff to improve surface water quality in the local karst terrain in Texas (ecoregion 255A). 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. Hubbard and Balfour (1993) warned that in karst areas highways may need to be located to avoid polluting surface waters that frequently end in underground caves in Virginia (ecoregion 231A). When investigating a cave system, they encountered raw sewage and petroleum fumes indicating the sensitivity of karst areas to pollution. Keith (1996) described extra precautions that were taken in Indiana to minimize the groundwater impact of a new road design in a karst area (ecoregion 222D).

Road dust can be a mechanism to transport unwanted chemicals to surface water. Christensen and others. (1997) observed recent accumulations of polycyclic aromatic hydrocarbons (PAHs) in a stream, and accredited dust from nearby roads as the source of the pollutant in a Wisconsin study.

Chemicals used to preserve utility poles and railway cross ties can be a potential source of pollution (Wan 1994). Wan (1994) found that concentrations of PAHs were excessive in the immediate vicinity of such sites compared to farm or range land. Concentrations of PAHs dropped rapidly from 550 μg/L to 23.2 μg/L within 13 feet. Background levels were between zero and 0.8 μg/L. Such findings emphasize the importance of maintaining vegetated buffers between rights of way and any sensitive water resources.

Polluted runoff can be treated in wetland areas that are natural (Mungur and others. 1994) or custom-designed (Barrett and others. 1998; Ellis and others. 1994; Karouna-Renier and Sparling 1997). Karouna-Renier and Sparling (1997) found that such treatment systems could remove up to 95 percent of metals, nutrients, and sediment before discharging to runoff water. 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 exfilration trench (Figure 3). 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). Sansalone and Buchberger (Sansalone and Buchberger 1995) found it to be effective in improving both rainfall and snow melt runoff from roads. Because of the wide range of runoff rates, multiple treatment methods may be necessary to decrease the pollutant load from large as well as small storms (Romero-Lozano 1995). Both detention basins for detaining the first flush of highly polluted runoff, followed by a filtration system to treat the runoff from later in the storm, which is likely to be at a higher flow rate, but require less treatment. As pollutants accumulate in treatment structures, future maintenance may be necessary to replace the polluted filtrate soil with fresh media before the structure becomes a source of pollution (Mikkelsen and others. 1995).

Sediment basins and similar structures built to contain polluted road runoff can themselves become a source of pollution through seepage into the ground, or through other forms of hydraulic or structural failure that may result in sediments with large amounts of adsorbed chemicals entering into the stream system. In some cases, the pollutants can become concentrated in these basins, increasing the risk of offsite pollution under low runoff conditions (Grasso and others. 1997; Morrison, Revitt, and Ellis 1995). Grasso and others. (1997) observed a lead content of 1392 mg/kg on one site, and recommended that soil washing be carried out to prevent offsite pollution. One of the best defenses against such risks is management through cleaning (Yousef and others. 1994) and maintaining such structures to minimize the risk of failure.

In more developed areas, past designs of runoff structures had tended to collect water and route it directly to a stream. New designs, which disperse water to ensure greater infiltration and onsite attenuation of pollutants, can improve runoff water quality (Li, Orland, and Hogenbirk 1998; Tysdal and others. 1997). 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; Pratt, Mantle, and Schofield 1995). Permeable pavement designs when combined with high-infiltration shoulders significantly decreased salt content in nearby groundwater (Church and Friesz 1993; Jongedyk and Bank 1999). Pratt and others. (1995) found that permeable pavement outflow levels of lead and suspended solids are significantly lower.

Reliability - The findings are generally reliable for their locality, but care needs to be taken in extrapolating to other conditions, particularly non-urban areas.

Ability to Extrapolate Findings - The water quality risks associated with hydrocarbon pollution are closely linked to the density of traffic. Those watersheds with minimal traffic are unlikely to experience any of the concerns discussed in the findings. Much of the research associated with chemical pollution from roads has taken place near large urban centers. Applying these results to more remote areas should only be done with extreme caution.

Watersheds with a high level of traffic are at risk for increased pollution either from the road surfaces, or from the mitigation measures that may have concentrated pollutants. As runoff and sedimentation events tend to be site-specific, managers should use caution in applying these findings to a specific watershed without some form of field validation and ongoing monitoring program.

Seconday Linkages - Hydrocarbons are linked either to runoff rates or sedimentation rates as discussed at the beginning of this chapter. Many of these pollutants are similar to those impacting large cities. Any technology that can improve the environment in a large urban area, will also improve the quality of water in the vicinity of a road.

In some cases, the sediment from a road may be high in calcium or a related cation, which could, in fact, neutralize acid rain. Under these conditions, chemicals leaving the road may improve the quality of water in the watershed.

Ability to address Issues - Much of the research done on chemical pollutants associated with roads has been carried out near urban centers. There is limited research on remote sites. Much of the more remote research work indicates that levels of pollution are measurable, but tend to be low.

Research Needs - There is a need to measure the levels of pollution risk from main roads which cross sensitive forest and grass land watersheds to determine the level of pollution from those roads. Generally, however, levels of pollution are low, and limited monitoring may be sufficient for a given watershed to ensure that the source of water is safe.

Fuels and other Contaminants from Accidental Spills

Issues and Risks - Accidents are rare on low use roads and other rights of way as found in many remote watersheds. Such accidents are related to the traffic density, quality of road, and likelihood of contaminant transport. Railroads also pose similar risks, particularly on aging lines, or on busy routes carrying industrial or petroleum chemicals.

Findings - 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 non mountainous terrain, but are seldom applicable to low traffic, remote watersheds. Chemicals that may be spilled include fuel, fertilizer, pesticides, and mining chemicals (USDA Forest Service 1998).

Airfields can often be sources of groundwater contamination from fuels due to spills or other activities (Levine and others. 1997).

Reliability - There is little information available the risk of accidental spills in remote areas. Whatever information can be found is likely to be site-specific, and judgement must be used to apply it elsewhere.

Ability to Extrapolate Findings - It is difficult to translate these findings to a specific watershed. The traffic levels and hazardous materials being transported are site-specific. 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.

Secondary Linkages - Accidents may occur anywhere along a given road or railroad, although stream crossing and bridges tend to be frequent sites of failure due to damage by floods, or a narrowing of the traveled way. Whether the pollutant is able to reach a nearby stream is an important concern. Frequently, transport of the pollutant overland, or through the soil, is dependent on the local climate, season, and hydrology. Mangers need to be aware of such a linkage.

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 the kind of accident that may occur near a water source.

Research Needs - 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 in risks associated with busier roads, so ongoing monitoring of those results for application to remote watersheds may be beneficial.

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 and ground water resources. In the past 15 years, there have been about 200 oil pipeline failures per year, with a net loss of about 600 barrels for each spill (U.S.DOT 1999) (1 barrel = 42 U.S. gallons).

Findings - Pipelines tend to have fewer accidents than other modes of transport (Figure 4) (Jones and Wishart 1996). Most modern pipeline systems are equipped with devices to quickly shut down pumping if a change in flow or pressure is sensed, to minimize pollution impacts of a failure (Ariman 1990).

Pipelines accidents may be the result of a number of causes (Figure 5). Road or construction accidents, or damage from boulders are common external factors that damage pipelines (Driver and Zimmerman 1998; Stalder 1997). Areas prone to severe erosion, landslides and earthquakes tend to have more accidents (Ariman 1990; Ballantyne 1995; Gray and Garcia-Lopez 1994; Hart, Row, and Drugovich 1995). For example, Hart and others. (1995) predicted that the probabilities of pipe 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, and 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 stream bank or bottom (Doeing, Bradley, and Carreon 1995; Teal, Carreon, and Bradley 1995). Soil shrinking and swelling, and freeze thaw cycles can lead to pipeline fatigue and premature failure. Corrosion due to electrolysis (Richardson 1996; Stalder 1997), and stress corrosion cracking can also occur on older pipelines (Wilson 1996). Above ground pipelines can suffer from wind fatigue leading to failure (Honegger, Nyman, and Nyman 1985). All pipelines may experience seam failure (Yaorong and others. 1996).

Risk assessment models have been developed (Hart, Row, and Drugovich 1995) which aid in the design stage. Other models have been developed to assist in identifying risk of failure during operation (Nessim and Stephens 1998). These models can be useful in identifying segments of pipe most at risk of failure, and ensure that the best maintenance strategies are applied to those segments. Risk rates of 0.0022 spills per mile per year are quoted in one Environmental Assessment (U.S.Bureau of Land Management 1978).

Table 7 presents the extent of contamination of 53 oil pipeline spills. The extent of a given spill depends on the pipeline characteristics and on the soils and terrain. Risks of failure from normal, predictive events can be reduced to almost zero with adequate pipeline monitoring (Stalder 1997). Technologies have developed to mitigate the impacts of spills quickly and effectively (for example, Sittig 1978; Southwest Oil World 1991).

Pipeline failures can also pollute groundwater. It is generally recognized that underground petroleum products tend to float on water, but the processes associated with breakdown of oil underground are not well understood (Revesz and others. 1996; Eganhouse and others. 1996). Revesz and others. (Eganhouse and others. 1996) focused their study on methane generation by anaerobic bacteria from oil from a pipeline leak. They observed that “substantial amounts of the volatile petroleum hydrocarbons are transported from the surface of the water table through the unsaturated zone as vapour, which subsequently dissipates to the atmosphere or is biodegaded.” Eganhouse and others. (1996) detected a plume containing aliphatic aromatic, and alicyclic hydrocarbons and identified the dominant underground breakdown process as microbial degradation.

In addition to oil pipelines, 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.

Reliability - These findings are generally reliable, as much of the pipeline industry receives close government scrutiny.

Ability to Extrapolate Findings - With pipelines, failures tend to be mechanical and predictable in nature. This means that the findings can be considered from an international database to apply locally.

Secondary Linkages - Erosion from pipeline installation is linked to the discussion on erosion sources at the beginning of this chapter. Landslide risk assessment may be linked to risk of pipeline failure, and may assist planners in locating pipelines in sensitive watersheds in more stable locations.

Underground storage tanks can leak, causing groundwater problems similar to those associated with pipelines (see Chapter 11). To minimize leakage risks, new underground installations generally have groundwater quality monitoring wells. Similar technology may be warranted for pipelines in sensitive watersheds.

Ability to Address Issues - With the current technology, we are able to address the risks associated with pipeline failure. Technologies to aid in managing pipelines to minimize failure are well established, as are controls to minimize pollution of surface and groundwater should a failure occur. Watershed managers with pipelines located in their watersheds should work with pipeline managers to ensure that practices available to minimize risks are in place to meet the risks on their watersheds (like earthquakes, erosion, etc.). In addition, the U.S. Dept. of Transportation has an Office of Pipeline Safety (OPS) to assist in addressing risks associated with pipelines. One of their responsibilities is to identify areas that are unusually sensitive to a hazardous liquid pipeline release, and they have an ongoing program that may assist watershed managers in risk management (email address given in references).

Research Needs - The oil industry has developed a high level of sophistication for managing their oil pipelines. There is a need to develop similar, but less costly technologies for application to water and sewer pipelines. 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.

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Sidebars

Sidebar 1. Offsite road erosion: A private timber industry company recently reconstructed a length of logging road. The engineers carefully graveled the segment of road they believed to be nearest to the stream about 300 ft downslope. When the winter rains began, however, they found that the greatest source of sediment was further up the road where the road crossed an ephemeral channel. The additional runoff from the road had resulted in downcutting of the ephemeral channel by nearly 1 ft, and delivering all of the detached sediment directly into the stream system. The graveled segment of the road had little runoff, and detached sediment was deposited between the road and the stream.

Tables

Table 11.1. Typical erosion rates for different types of land use in the U.S.

Location and Ecoregion	Surface Cover	Erosion rate (t/acre/yr)	Reference
Eastern Watersheds M221	Forests	0.003 – 0.32	Patric 1976
Appalachian Trail M221	Trail	4 – 60	Burde and Renfro 1986
Southeast M221D	Roads	5 - 144	Swift 1984 a and b and 1988
Southern Watersheds (various ecoregions)	Forest Meadow Prescribed burn Careless clear-cut Roads	trace – 0.32 0.06 – 0.1 0.01 – 0.23 1.35 16 - 150	Yoho 1980
Central Arkansas M231C	Roads	6.8 – 33.7 4 – 38.5	Beasley and others. 1984 Miller and others. 1985
SE Oklahoma M231A	Roads	8 – 77	Vowell 1985
Western Watersheds	Range land	0.1 – 1.8	USDA 1989
Northern Rockies M333D	Forests Forest Watershed Undisturbed With roads	0.04 0.0 0.02	Megahan 1974, McClelland and others. 1997 Megahan and Kidd (1972b)
	Roads	7.5 - 22	Ketcheson and Megahan 1996; Megahan and others. 1986; Megahan and Kidd 1972a
Washington Olympics M242A	Roads	46 – 550	Reid and Dunne 1984
Oregon Cascades M242B	Forested Watershed Roads added Harvested Roads	0.11 0.56 18.4 0.22 - 24	Fredrikson 1970 (Most of road and harvest erosion attributed to landslides) Foltz 1996
Oregon Coast Range M242A	Roads	1 - 22	Tysdal and others. 1997
Oregon Coast	Forests	0.4	Beschta 1978

M242A			
Northern California Coast Range watershed M261B	Undisturbed forest After roads After roads and logging	0.008 0.63 1.9	Rice and others. 1979

Table 11.2. Observed and predicted erosion rates and sediment plume lengths below forest roads using the WEPP:Road program (Elliot, Hall, and Scheele 1999)

Site and Ecoregion		Source
Erosion Rate	(t/acre/yr)	
Observed at Zena Creek, ID M332A	7.9	Megahan and Kidd, 1972a (Included entire new road prism)
Predicted for Zena Creek by WEPP:Road	4.5 and 5.4	Assumed 10 percent road gradient and Warren, ID climate for road traveled way and ditch
Observed bare and graveled road in Southern Appalachians M221D	5.0 to 68	Swift 1984a and b
Predicted for Southern Appalachians by WEPP:Road	6.8 to 83	Road dimensions from publications and Cullowee, NC climate
Observed for Alum Creek, AR M231A	6.8 – 33.7	Beasley and others. 1984 (Vegetated fill slope and ditch)
Predicted for Alum Creek, AR	19 - 49	Assumed crowned road and two ditches
Observed bare and gravelled road for Fernow NF, WV M221B	6.0 – 52.5	Kochenderfer and Helvey 1987
Predicted for Fernow NF, WV	16 – 37	Road dimensions from publication, used Clarksburg,WV climate
Sediment Plume Length	(feet)	
Observed on Silver Creek Watershed, ID M332A Cross drain Below fill	35 - 602 (mean = 163) 1.3 – 217 (Mean = 12.5)	Ketcheson and Megahan 1996
Observed in Nez Perce NF, Central ID M332A Below culverts	80 percent less than a Mean of 78	Wasniewski 1994
Predicted for Silvercreek Cross drain Below fill	101 0	Average dimensions in publication Used Deadwood Dam, ID climate, assumed road gradient was 10 percent
Predicted for Wine Springs Watershed, NC M221D	L = 5.1 + 0.00197 M => 74 ft	McNulty and others. 1995 (Length in meters and M is predicted sediment yield from road) Length with a WEPP-Road predicted yield of 9979 kg
Filters strip widths in SE U.S. M221D	43 + 1.4 x percent slope = 85 ft	Swift 1986

Predicted for Wine Springs Watershed, NC	118	For Cullowhee, NC climate and a 330-ft road length, 10 percent gradient, 30 percent buffer gradient
Observed in Tuskegee NF, Alabama M231B	164 - 197	Grace 1998, occurred Aug ’97 – Jan ‘98
Predicted for Site	131	Assumed Opelika Climate, 200 ft road, 5 percent gradient, 9 percent buffer gradient

Table 11.3. Distribution of land and sediment sources in a mixed watershed in Idaho (ecoregion 331A) (ID DEQ 1997)

Land Use	Percent of Area	Percent of sediment
Agriculture	60.5	81
Forest	14.2	7
Pasture	-	8.6
Urban	16.6	5
Roads	Estimate 1	8

Table 11.4. Effectiveness of erosion mitigation techniques

Condition	Reduction (percent)	Reference
Erosion mat Seeding	74 – 99 82 - 95	Grace and others. 1996 (ecoregion 231B)
Grass on fill slope	46 – 81	Appelboom and others. 1998 (ecoregion 232C)
Straw and asphalt tack or erosion mats (Depends on percent cover) Straw	60 – 100 60 - 80	Burroughs and King 1989 (ecoregion M332A)

Table 11.5. Pollutants that have been observed in runoff from road surfaces

Pollutant	Comment	Reference
Cd, Cu, Pb, Zn	Treated in wetlands	Mungur and others. 1994
Aliphatic hydrocarbons, polycyclic aromatic hydrocarbons (PAHs), phenols, 4- and 5-ring PAHs	Extracted from sediment contaminated with road runoff	Boxall and Maltby 1995
Road salt	Mobilized mercury in soil	MacLeod and others. 1996
Cd, Zn, Cu, Hg, Ni, Pb, PAHs	High concentrations in lake near highway	Baekken 1994
Highway deicing salt	Na adsorbed in soil	Shanley 1994
Cu, Zn, hydrocarbons, PAHs	Accumulated in aquatic biota	Maltby and others. 1995
PAHs	Altered aquatic communities	Boxall and Maltby 1997
2,4-D and picloram	Applied to road ditches, leached into groundwater	Kohl and others. 1994
Na, Cl, Ca	In deicing and dust control chemicals	Kjensmo 1997
Ca, Mg, Na, K, Cl	Captured in peat bogs	Pugh and others. 1996
Total petroleum hydrocarbons (TPH), Pb, Zn	Reduced by vegetation	Ellis and others. 1994
Nitrophenols	In solution	Wuest and others. 1994

Pb, PAHs Cd, Cu	Attached to sediments Attached to organic carbon	
Heavy metals, petroleum hydrocarbons, pesticides, sediment, nutrients	Treatment ponds can remove up to 95 percent of pollutants	Karouna-Renier and Sparling 1997
Solids, metals, hydrocarbons, herbicides, de-icing agents	Offsite impacts immediately downstream of discharge	Ellis and others. 1997
Heavy metals, inorganic salts, aromatic hydrocarbons, suspended solids	Accumulating in a road infiltration system	Mikkelsen and others. 1995

Table 11.6. Mean concentration of a number of pollutants in highway runoff in Minnesota (Thomson and others. 1997)

Pollutant	Range (mg/L)	Mean (mg/L)
Total Nitrogen	0.6 – 8.14	1.67
Chloride	1 - 46,000	1802
Sulphate	5 - 650	45
Sodium	2 – 67,000	3033
Total phosphorous	0.06 – 7.8	0.6
BOD	1 – 60	12.6
COD	2 – 3380	207
Total suspended solids	8 – 950	118
Total dissolved solids	22 – 81,700	10,440
	(µg/L)	
Chromium	1.5 – 110	13
Copper	3 – 780	47
Iron	180 – 45,000	4162
Lead	11 – 2100	207
Zinc	10 – 1200	174
Nickel	1 – 57	10
Cadmium	0.2 – 12	1.7
Mercury	0.08 – 5.6	0.49
Aluminum	30 – 14,000	2694
Arsenic	0.1 – 340	19

Table 11.7**.** Extent of soil contamination by 53 oil spills of various sizes in Alberta, Canada (Mackay and Mohtadi 1975)

Average volume (barrels)	Average Area (sq ft)	Average film thickness (in.)
54	8000	0.4
880	70,000	0.8
13,200	600,000	1.6

Figure 11.1**.** Surfaces and flow paths associated with a road cross section.

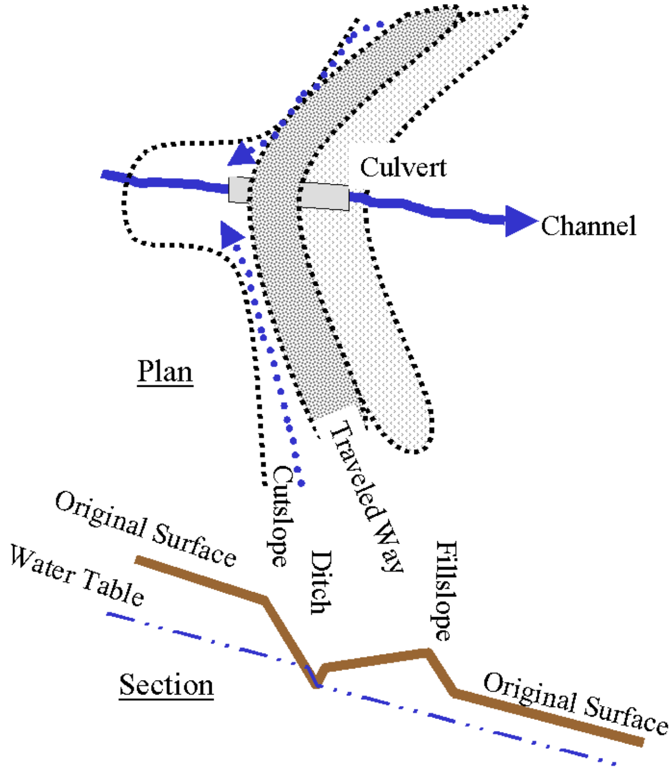


Figure 11.2. Interacting factors that impact soil erosion

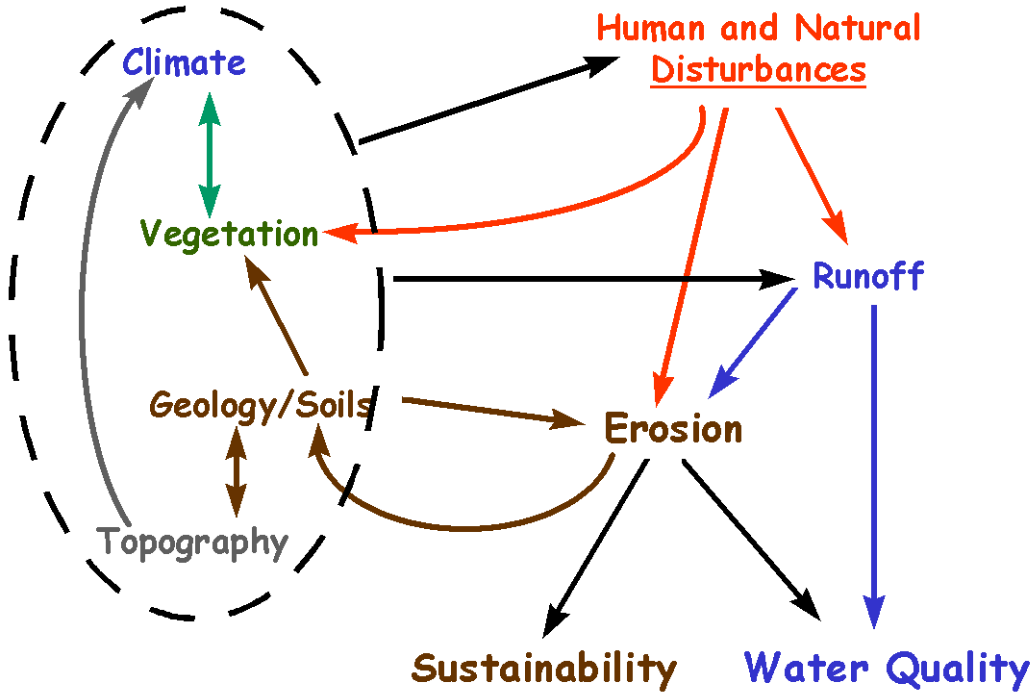


Figure 11.3. Diagram of a partial infiltration trench

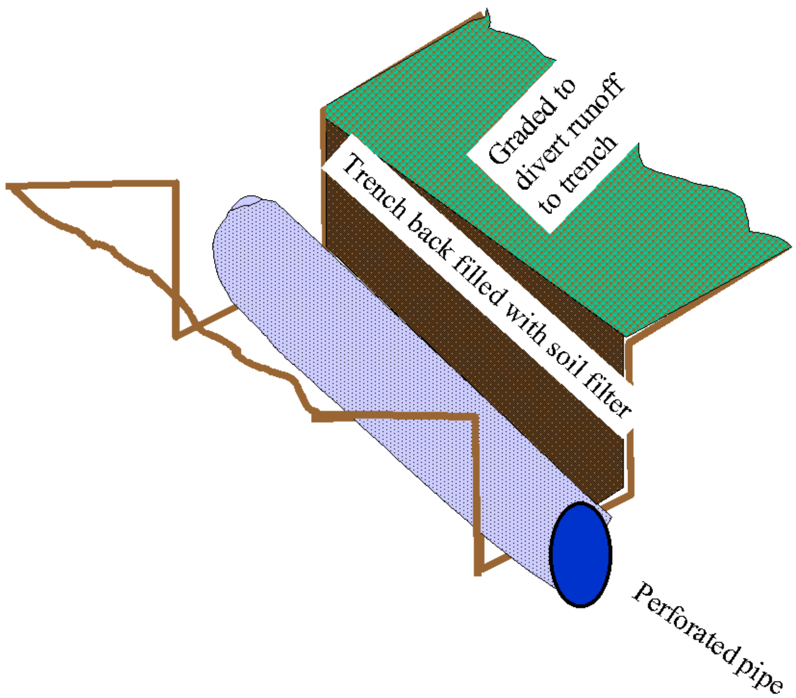


Figure 11.4. Comparison on annual injuries for various modes of transportation (Jones and Wishart 1996)

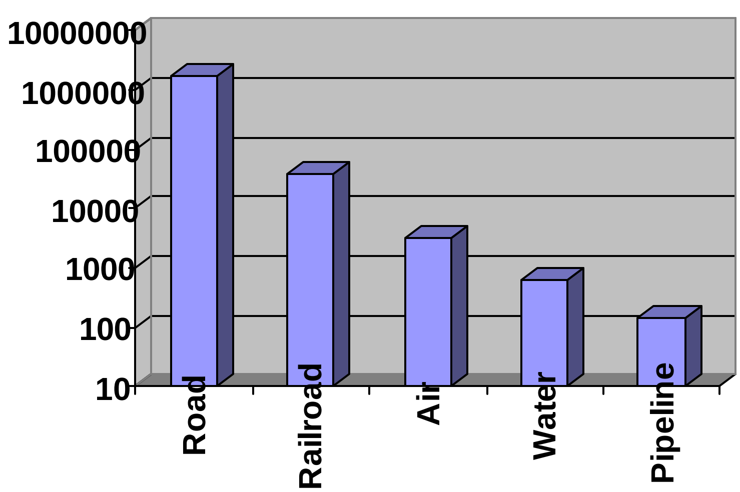
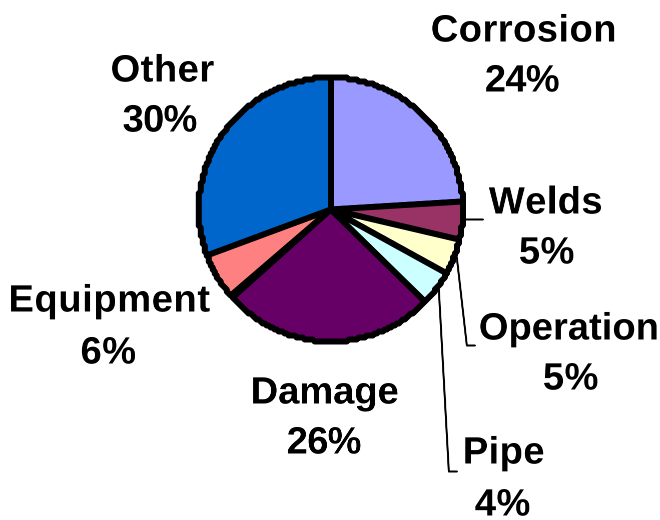


Figure 11.5. Summary of causes of pipeline accidents in 1998 in the U.S. (U.S.DOT 1999)



Acronyms and abbreviations

µg	Migrogram = 10^{-6} gram
DOT	U.S. Dept. of Transport
L	Liter = 0.2642 U.S. gallons
mg	milligram = 10^{-3} gram
OPS	Office of Pipeline Safety in the U.S. Dept. of Transport
PAH	Polycyclic aromatic hydrocarbons
WWW	World Wide Web

Glossary Terms

Cutslope	Excavated slope uphill from a road located on the side of a steep hill (fig. 1)
Erosion	
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 (fig. 1)
Karst	Geologic formation in limestone strata containing numerous dissolved channels resulting in high hydraulic conductivity and high risk of groundwater pollution
Pollution	
Sedimentation	
Travel Way	The surface of a road on which traffic travels
Watershed	

Chapter 12

Past Land Use

Karen Solari

Introduction - Public lands, such as those administrated by the Forest Service, serve multiple uses including grazing, mining, oil and gas production, landfilling of municipal wastes, and timber harvest. These activities are conducted by private parties and state and local governments through permits, leases, special use authorizations, and other types of licenses. In addition, the Federal land managers construct roads, campgrounds, and administrative sites and conduct waste generating activities such as wood treatment, pesticide application, vehicle maintenance, and painting.

A review of forty-two State reports in relation to public ground water supplies indicated that the most frequently identified contaminant sources were underground storage tanks, septic tank systems, surface impoundments, municipal landfills, agricultural activities, abandoned hazardous waste sites, regulated hazardous waste sites, land application/treatment, injection wells, and on-site industrial and other landfills. (Canter and Maness, 1995). Many of these types of activities have been conducted on public lands or may have been conducted on prior of the land. In addition, because of remote locations and easy access, the public lands are attractive targets for illegal dumping of hazardous waste.

In many cases, even if properly maintained and operated when active and closed in compliance with the regulations existing at the time, activities which were discontinued several years ago may present current problems. This is compounded if the activity was not operated or closed properly. This chapter discusses the drinking water impact of past land uses. Determining the impact of past land use on drinking water is a three-fold problem:

- First, the past land use must be determined. This is not always readily apparent and may require record searches.
- Second, the potential impacts of these uses on drinking water must be identified.
- Finally, because some pollutants spread quickly, contaminating groundwater and surface water several miles from the site past land uses of adjacent properties should also be considered.

Hazardous Materials

Issues: Identification of past uses of hazardous materials may require an extensive records review in addition to field work. Sources of information for historic land uses include:

- Land Management Agency files relating to the use, storage or disposal of hazardous substances and petroleum products special use permits and leases or reports of a release and any cleanup activities.
- Federal, state, and local regulatory agencies responsible for permitting and monitoring environmental and human health concerns
- Databases maintained by EPA and states that identify sites of known or suspected contamination such as:
  - list of known or suspected abandoned hazardous wastes sites

- list of Leaking Underground Storage Tanks
- list of hazardous waste generators and hazardous waste transport, storage and disposal facilities

· Aerial photographs, particularly historic aerial photos
· Topographic and county plat maps.

Findings: Once the past hazardous materials activities are identified, possible areas of contamination or potential contamination can be determined. Common types of past hazardous materials activities on public lands and types of contamination associated with these activities include the following.

Landfills: Under federal land management agency regulations and policies, numerous landfills have been permitted to operate on public lands. Those in operation prior to 1970's were not regulated as strictly as today. Due to lack of regulation; old, abandoned and closed landfills may contain volatile organic compounds, pesticides, polychlorinated biphenyls (PCBs) from electrical transformers, polynuclear aromatic hydrocarbons (PAHs), cyanides, heavy metals, and other contamination. Even landfills accepting strictly domestic or household waste can contain oil and grease, paint, corrosives, solvents and other miscellaneous consumer chemicals. (EPA, undated).

The fact that a landfill is covered with healthy vegetation does not eliminate the possibility of a problem. Rainfall seeping into the landfill or groundwater moving through the landfill can come into contact with buried waste, pick up contaminates, and carry them several miles from the site. High groundwater tables, permeable soils, poor cover materials contribute to increased potential for hazardous materials leaching from landfills into ground or surface water.

Abandoned and inactive mines: The reader should see Chapter 8 for a detailed discussion of mining operations. The environmental problems which can be caused by mining include acid mine drainage, heavy metal runoff from waste piles, cyanide escaping from leaching operations, and failure of surface impoundments and tailings dams. These problems are all compounded when the mining operation is no longer active and has no one monitoring day-to-day conditions. The USDA Office of Inspector General estimates that there are over 38,000 abandoned and inactive hard rock mines on national forest system lands. An inventory of abandoned and inactive coal mines is being developed.

Underground Storage Tanks (USTs): Environmental Protection Agency (EPA) has received more than 300,000 reports of releases from UST systems and receives an average of 30,000 new reports each year. EPA has identified gasoline as one of the most common sources of groundwater pollution. (EPA, 1998). In 1986, EPA published regulations with the goals of preventing and cleaning up releases from USTs. These regulations (40 CFR Part 280) require that USTs which contain hazardous substances, including fuels, be removed by December 22, 1998 or have spill, overfill, and corrosion protection. The regulations also require that installation and closure of USTs be registered with the State or EPA. To comply with these regulations, the Forest Service has removed over 1,600 USTs and has initiated several projects to cleanup groundwater contaminated by leaking USTs.

Records should be available through the State or EPA identifying where USTs are located; where cleanup operations are ongoing; and where USTs have been removed. The possibility also exists that USTs may be present and not registered with the appropriate agency. During field visits, look for indications of former structures or operations on the property. Note the presence of partially exposed, capped, or uncapped pipes. These may indicate the presence of monitoring wells or injection wells, or serve as vent pipes or fill pipes for an underground storage tank. On property where motor vehicles were operated regularly, be skeptical when there is no apparent refueling source. There is likely to be an underground storage tank present. (Forest Service, 1999)

Illegal dumps: Illegal dumping is disposal of waste in an unpermitted area. This is usually done to avoid disposal fees or the time and effort required for proper disposal. Dumped materials typically include nonhazardous wastes such as scrap tires, bulky items, and yard waste but can include hazardous wastes such as asbestos, household chemicals and paints, automotive fluids, and commercial or industrial wastes. Because of remote locations and easy access, public lands are particularly vulnerable to illegal dumping. If not addressed, illegal dumps often attract more waste. The potential for contaminated runoff and groundwater contamination exists due to the uncontrolled nature of these dumps.

Shooting Ranges: Abandoned shooting ranges pose the potential for lead contaminates entering ground and surface water. Acidic rainfall or acidic soil can dissolve weathered lead compounds. Shooting ranges located in areas with these conditions have an increased potential for transporting contaminates offsite. Ranges which do not have good ground cover also pose a higher risk of lead contamination migrating offsite. (Sever, undated).

Wood Treatment. Wood posts have been treated in dip tanks at various field locations throughout Forest Service lands. The wood preservatives used in these dip tanks included creosote, pentachlorophenol (PCP), and a chromium, copper and arsenic compound (CCA). The common practice was to dip wood posts then move them to a drip or drying area. The Forest Service has discontinued these treatment operations and cleaned up all known sites. There is the potential for both ground and surface water contamination from any sites which have not been cleaned up.

Formerly Used Defense Sites (FUDS): The Department of Defense (DOD) has the responsibility for the cleanup of past military operations conducted on public and private lands. DOD estimates there are over 9,000 contaminated sites ranging from military training sites to industrial facilities which must be cleaned up. These sites pose a wide range of environmental hazards including unexploded ordinances from the training sites and contamination from the solvents, fuels, and organic and petroleum compounds used in the industrial facilities. (COE, undated). Many of these 9,000 formerly used defense sites are on public land. For example, the Forest Service has identified over 100 FUDS on national forest system lands.

Reliability of Findings: The information on types of hazardous material activities and the contaminates associated with these activities is reliable because extensive site specific data has been compiled through the Forest Service and the EPA hazardous waste site cleanup programs.

Ability to Extrapolate Findings: Much of the information on abandoned hazardous materials activities is site specific. Local conditions such as climate, soils, and hydrology must be considered along with the type of hazardous materials activity.

Secondary Linkages: See Chapter 8 for additional information on mining operations.

Ability to Address Issues: The technology to address the cleanup of most types of hazardous waste sites is available. The scope of the problem and the cost of the cleanups have made this a multi-year program. Within the Forest Service, it is estimated that the abandoned hazardous waste sites will require about $2 billion and 50 years to cleanup.

Research Needs: Research is ongoing to develop effective, cost efficient methods for cleaning up sites and the surface and groundwater these sites have contaminated..

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Abandoned Roads, Railroads, and Pipelines

W. J. Elliot

Sediment Error! Bookmark not defined.

Introduction - Chapter 10 discussed the impacts of roads and similar disturbances on the runoff and sediment yield from watersheds. When such features are abandoned, they can continue to be sources of sediment through chronic surface erosion or episodic mass failure (Elliot and others 1996). The compacted nature of a disturbed surface frequently restricts vegetation regrowth. Bare surfaces are more susceptible to erosion, and steep areas without trees are more susceptible to landslides (Hammond et al. 1992). In some cases, local freeze/thaw processes or minor slumping of fill shoulders can collect surface water leading to saturation of the fill and an increased risk of mass failure (Sidebar 1). In either case, both surface erosion and mass failure can lead to higher sediment loads in surface water (McClelland and others 1997).

Findings – Limited information is available related to the impacts of abandoned roads, railroads, and pipelines. Generally, the same principles that apply to active features can also apply to those that have been abandoned (See chapter 11).

There are several general principals that can be applied to analyzing and mitigating potential sediment sources from abandoned structures. 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 similar to mine reclamation (see chapter 8). To encourage infiltration and revegetation, it may be necessary to discourage off-road vehicle traffic by installing some permanent barrier or removing a length of the road.

Surface erosion rates drop significantly when roads are closed. Figure 12.1 shows findings by Swift (1984) on the relative impacts of differences of road surface during the first two years of abandonment, compared to rates erosion rates during construction and logging. In the Oregon Cascades (Ecoregion

M242B) Foltz (1996) observed that erosion rates dropped from 4 to 0.5 t/acre or 20 to 2.5 t/acre the first year of closure, for marginal and good quality aggregate respectively. In both the Swift (1984) and Foltz (1996) cases, erosion would likely continue to drop to background levels as the density of vegetation on the road surface increased.

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

Poor drainage can lead to saturation of road prisms and mass failure as well. In steep terrain, abandoned roads that do not shed surface water can become saturated, increasing the likelihood of failure (Sidebar 1).

In an effort to reduce the impacts of such failures, many government management agencies are removing unwanted roads. In many National Forests, watershed restoration is synonymous with road removal (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, stability, and revegetation when planning any road closure or obliteration activity. Table 1 provides a summary of the management options for abandoned roads. Elliot and others (1996) warn that the disturbances caused by road closure may cause more erosion than from a road that has become revegetated and is hydrologically stable.

Reliability of Findings - There has been little research on abandoned roads, as they are perceived as a management issue rather than a research issue. Consequently, much of the information published about abandoned roads tends to be site-specific, focusing on methods rather than analysis (Moll 1996).

Mass failures on active and abandoned roads are frequently not evaluated until after they occur, so caution should be used in interpreting the reports of such failures. Report details usually focus on the failure, while they ignore information on the road design details and an evaluation from many miles of nearby road which did not fail (McClelland and others 1997; Megahan and others 1978). This makes it difficult to determine either the factors that led to failure, or the associated risk of a similar failure in a given area.

Ability to Extrapolate Findings - Most documented problems associated with abandoned roads and other structures tend to be site specific. However, there have been sufficient problems to realize that blocked culverts, concentrated flow erosion, and mass failure are common. Any efforts to minimize these processes will reduce the problems of sedimentation associated with abandoned roads and railroads.

Secondary Linkages - Abandoned roads and railroads can provide corridors for illegal dumping of chemicals in remote areas. They can also become passages for recreation vehicles, hunters and fishermen (Chapter 7). They may be areas for grazing by wildlife, as grasses may be sown, or may naturally be the first type of vegetation to grow on abandoned sites.

Previous erosion from roads or other major watershed disturbances may have increased the amount of sediment stored in a stream system. Even though the disturbance may be stable, it is possible that sediment resulting from that disturbance may still be working its way through the stream system. For example, channel observations following the landslide described in Elliot and others (1994) indicated that the channel was aggrading rather than degrading where landslide debris had crossed the stream channel (Ecoregion M333). In a watershed in Texas (Ecoregion 231), a major headcut was observed to be slowly working its way upstream, contributing sediment to a sensitive lake. It is likely that the source of eroded sediment was from watershed disturbances many years ago, and the headcut process may have been initiated decades ago with channel disturbances far downstream. It will likely be many years before a new geomorphic equilibrium is established in either of these watersheds.

In some instances, abandoned roads or similar features adjacent to stream channels can cause the development of point bars, which deflect the flow to the opposite side of the channel. The net effect of the structure is increased channel erosion.

The disturbance caused by removal of roads and similar features can also be a source of erosion. In sensitive watersheds, sediment reduction practices similar to those for road construction, like silt fences, mulching or reseeding, should be applied to removal activities (Moll 1996).

Ability to Address Issues - The technology to remove abandoned roads is well established (Moll 1996). Numerous agencies including the Forest Service (Moll 1996; USDA Forest Service1989) and the National Park Service (Spreiter 1999) have specialists to provide technical assistance in road closure, stabilization, revegetation and removal.

Many abandoned roads and railroads tend to require site specific considerations in reclamation. The same level of design that went in to creating some of these sites may be required to mitigate them (Elliot and others 1996). Although this design expertise is available, the cost may be prohibitive.

Research Needs - Reports related to problems associated with abandoned roads and railroads focus specifically on culvert or mass failures, rather than on the entire road network and other types of erosion and sedimentation processes. These other processes, such as 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. Research is also needed to determine risks of failure or erosion for specific road networks. From such studies, 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 failure which may impact offsite water quality as well as other resources. 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.

Soils Polluted with Fuels or Other Chemicals

Past activities within watershed may have included railway yards (Welsh and others 1993), fueling depots, saw mills, airstrips (Levine and others 1997), or other features which polluted the soil. Also, abandoned roads may have provided access for illegal dumping of chemicals in concentrations that are still high enough to potentially impact the quality of surface or ground water.

Issues and Risks - Past activities like rail yards, mill sites, or vehicle-fueling stations may have polluted the soil with fuels or other chemicals. Underground fuel tanks may have been abandoned with fuel still in the corroding tank, or may have polluted the soil, which now is a potential point source of watershed pollution.

There have been numerous cases where chemicals have been intentionally dumped near roads. The roads may or may not be abandoned, but the chemicals may pose a threat to the quality of both surface and ground water.

Findings - Fuels that have been spilled may migrate over a number of years from the site. Frequently these sites may be in developed areas, which are undergoing redevelopment such as former railway yards (Welsh and others 1993). Site specific studies are generally required to determine the migration potential of the fuel and its chemical derivatives, and the risk to health associated with migration into potential groundwater sources.

In rare cases, roads or other structures have been built or reconstructed on sites that were already polluted. Identification and remediation of these sites can be expensive. In many cases it is not possible to remove the structure or road, so the site must be treated by the installation of horizontal wells beneath the structure (Griffin 1998). It is generally much easier to clean up a polluted site before constructing a road, runway, or building than after.

Areas of deposition around abandoned roads or other structures may contain elevated levels of pollutants. Mikkelsen and others (1995) found that levels of concentration, however, decreased rapidly to background levels in less than 1.5 m. Groundwater risks are limited, but such sites may be sources of surface pollution from future construction activities should a polluted site begin to erode.

Roads were found to be a source of polycyclic aromatic hydrocarbons that were generated from coal-wood gasification plants that have been closed for decades in Wisconsin (Christensen and others 1997). In their study they found that the PAHs were transported from the roads to the surface water by dust rather than water erosion. Similar concerns are also under investigation in a current study in which the author is collaborating on dirt roads in an area contaminated by radioactivity at an old munitions factory on Colorado rangeland.

Reliability of Findings - Most information on mitigation efforts is reliable because of careful scrutiny of the mitigation by state and federal agencies.

Ability to Extrapolate Findings - The results of pollutant investigation studies are often difficult to apply to other sites because of the site-specific nature associated with individual cleanup efforts. Special attention needs to be given to the climate, soils, and hydrologic aspects of an individual site to determine if it can be applied to another site.

Secondary Linkages - There has been a major effort throughout the U.S. to replace aging underground storage tanks (USTs) common in service stations and depots. Much of this information may help in evaluating abandoned sites that have had major fuel spills.

Ability to Address Issues - Technology is well established to treat soils contaminated with fuels, and for solving problems associated with leaking USTs. For abandoned sites with lower concentrations of contaminants, there may be simpler solutions, however, depending on site conditions (e.g. Welsh and others 1993).

Sites polluted with unknown chemicals require specialists to evaluate and mitigate a given problem. Although the technology is generally available, it will be expensive to apply in remote areas accessed by low volume or abandoned roads.

Research Needs – Because of the diverse nature of potential contaminants, problems will be site-specific, and there is no broad-scale research that could be initiated to address this issue.

Road Excavations Creating Acid Drainage

Past road construction, particularly in areas to access mineral resources may have exposed subsoil material that has the potential to create acid drainage (See chapter 9).

Issues and Risks – Chapter 9 describes the processes that lead to the creation of acid mine drainage. On roads directly associated with mining, mitigating such conditions would likely be included with other remediation work associated with the mining permit. On older roads, particularly those that have been abandoned, the watershed manager may be unaware of acid drainage problems, and there may be no organization to take responsibility for mitigation.

Findings and Reliability of Findings – See chapter 9

Ability to Extrapolate Findings – Treatment of acid drainage is well established for mining sites. Similar technology can readily be applied to road sites once they have been identified.

Secondary Linkages – Some researchers have found that surface aggregate may generate sediment fines that can buffer acid rain effects (MacLeod and others 1996). Similar buffering may also reduce impacts of acid mine drainage from abandoned roads.

Ability to Address Issues – Mitigation of acid drainage from road cuts would be similar to methods used in mining. The cost, however, may be prohibitive.

Research Needs – Research on managing acid mine drainage is ongoing, and will likely be adequate to address similar problems on active or abandoned roads (See Chapter 9).

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Table 13.1. Management options for abandoned 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	
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 butresses to stabilize cut and fill slopes	Moll 1996
Remove, “recontour “ or obliterate road prism	Expensive ($0.60 to $1.50 per lineal foot (Moll 1996)) Increase revegetation rate by excavating until the old topsoil is reached (Spreiter 1999)
Mitigate obliterated road prism with slash, mulch, geotextile or seeding	Moll 1996

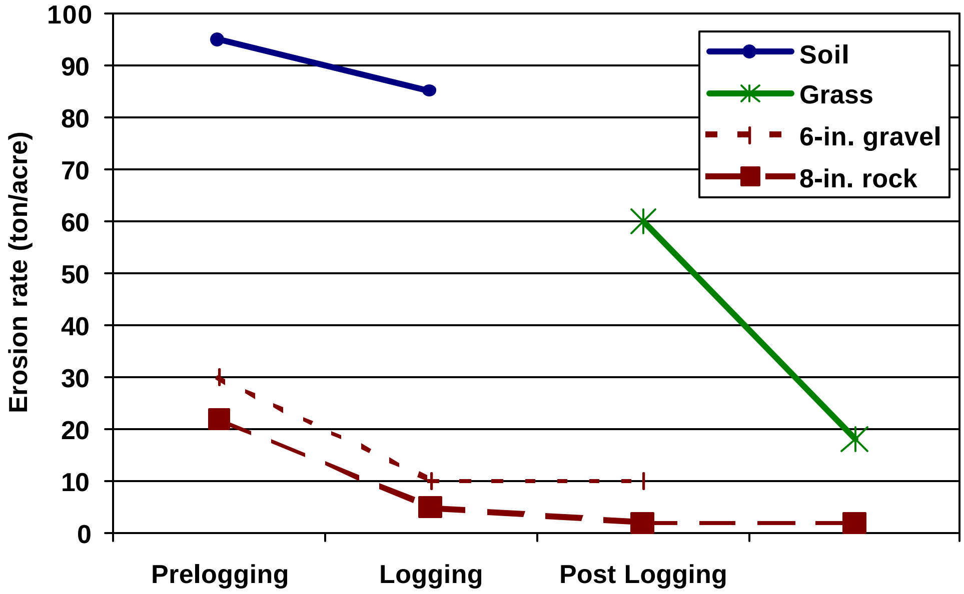

Figure 13.1. Mean soil loss rates for three road surfaces before, during, and after logging in Ecoregion M221D (based on Swift 1984).

Sidebar
Sidebar 1. In the mid 90s, there were two major road failures on a small road network (approximately 5 miles) of road in the Nez Perce National Forest. The slope steepness was 40 to 50 percent. A private owner downstream of the forest had the intake to his hydroelectric system blocked on two occasions with excessive sediment. An inspection of the road network by a group of engineers determined that the source of the sediment was from two separate mass failures on the road network. In both cases, surface runoff was collected by the road and was transported for several hundred feet until the road flattened. At that point, the runoff saturated the road prism, and the road failed. Below the road, a small debris torrent was initiated, which resulted in a large volume of sediment carried all the way to the stream system some 300 feet below the road. The forest decided that complete removal of the road was preferable to simply sloping the road away from the hill to shed water.

Acronyms and abbreviations

UST Underground storage tank

Glossary Terms

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 shape.
Rock Buttress A thick layer of rock placed on top of a steep sideslope to reduce the risk of a mass failure (Moll 1996)
Slumping A mass failure generally due to an increase in the water content within the soil profile.
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.

Abandoned Wells

Issues: Abandoned wells are those whose use has been permanently discontinued for reasons of nonproduction, nonuse or disrepair. These wells may or may not have been properly closed or plugged. It is the improperly abandoned wells which are of concern since they can serve as conduits for migration of contaminants to aquifers and between aquifers, which may be underground sources of drinking water.

There are numerous types of abandoned wells on public lands. Those associated with natural resource management are mineral exploration wells, oil and gas production wells, and stock watering wells. Those associated with administrative and recreational developments are water wells used for drinking water supplies or irrigation, and disposal wells used for stormwater runoff or in vehicle shops. Septic systems are also considered disposal wells when industrial or commercial wastes are treated along with sanitary wastes (40 CFR Part 146).

Although Federal, State or local regulations address proper closure of abandoned wells for all the above categories, this does not mean that all abandoned wells on public land have been closed or plugged as directed. Many of the wells were developed at a time when these regulations did not exist. Other wells have been abandoned temporarily, to allow for further development 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 (EPA, 1998).

Certain States have already banned such dry wells and RCRA related cleanups required due to contamination at such sites.

The number of abandoned wells on public land is unknown. For example, the Forest Service does have inventories of some categories of in-use wells, but not of abandoned wells. Knowledge of number and location of such wells would be limited to District staff, in most instances, and 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 groundwater protection (Nye, 1987). 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 groundwater protection.

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

Field studies have shown that underground water sources can be significantly degraded with organic and inorganic contaminants due to storm-water runoff and dry wells in automotive shops (Ogden, 1987). In certain geologic formations, abandoned water wells are prone to collapse and, when drilled through multiple aquifers, can cause significant problems (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 groundwater pollution including salinization (Gass, 1988). Even plugged boreholes may have compromised integrity, allowing pollutant transport; EPA is currently funding a study of groundwater threats from improperly abandoned boreholes (Kubichek, 1997).

Reliability of Findings: Proper well closure is heavily regulated at present, but not heavily enforced. The Forest Service has limited knowledge of the number and location of abandoned wells that have not been properly closed or plugged. Their existence does not mean groundwater contamination will occur; only that is has the potential to occur.

Ability to Extrapolate Findings: Wells on public lands 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 development, the more likely the well was drilled and closed properly. Forest Service dry wells in vehicle shops may not have the extent of waste disposal as would a commercial facility, but the pollution potential still exists. The hydrogeologic formations of some Forest areas could contribute to a greater potential for contamination. There could be a greater problem with abandoned oil and gas wells in the Allegheny and Appalachian mountains, where wells are more numerous and older than in other Forest areas.

Secondary Linkages: None

Ability To Address Issues: The issue of abandoned well closure is defined well in Federal, State and local regulations. The extent of the problem on public land is not known since there is no inventory of abandoned wells.

Research Needs: Local land managers will need to survey or inventory existing abandoned wells and review closure methods, if any, in order to quantify the work which must be done to eliminate this potential contamination problem.

Definitions

Abandoned well -any well which is unusable, or which is in such disrepair that its continued use is impractical or is an environmental, safety or public health hazard.

Closed well - a well that has been permanently disconnected & filled such that contamination cannot move from the surface into the aquifer.

Drywell - a well, 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.

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.

Well - a bored, drilled or driven shaft, or a dug hole, whose depth is greater than the largest surface dimension.

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Sediment Transport In Streams

Introduction -In many regions of the country, streams are currently,transporting sediment from past land uses and management practices. Sediment from previous poor agriculture, logging practices, roads, site preparation, and other land uses as well as sediment from past catastrophic events such as wildfire, large storms and landslides enters streams and is transported over time downstream. The rate of transport depends on the size of sediments, gradient of streams, and streamflow.

At many locations, sediment from past erosion is influencing present day channel conditions and sediment transport. Forests were cleared for grazing, mining, and agriculture in the 1800's and early 1900's. To illustrate, 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 excess sediment supply to streams exceeded the stream's ability to transport causing deposition. Huge volumes of sediment were deposited in the stream channels and flood plains. The eroding lands were eventually abandoned and reverted to forest or were planted to trees and pasture under conservation programs in the mid-to-late1900's. The landscape was stabilized and sediment yields to streams were greatly reduced. Because the runoff from the landscape carried little sediment, the streams had more available energy to transport sediment and began transporting sediment out of storage. The streams started to down-cut and head-cut up stream through the stored sediment. The process continues today in many river and tributary systems. For these streams, much of the sediment being transported today is from abandoned land uses and in addition to that from natural sources and from current watershed activities.

Issues and Risks – Issues and risks resulting from sediment transport from past and abandoned land uses include: (1) sediment yields from a watershed may be higher than expected given the present forest and grass land management, (2) streams liquidating stored sediment often have unstable stream channels and banks, and (3) stored sediment from past mining, agriculture, and other land uses may have harmful chemicals and metals attached and impair water quality.

Findings - Small sediments < about 0.06 mm (silt) size tend to move relatively rapidly through the channel system as wash load. These fine sediments are the primary factor influencing turbidity. Larger sediments move as bed material load and can have short to long residence times in the channel system depending on particle size. Bunte' and MacDonald (1998) made a comprehensive review of the literature dealing with sediment transport distance as a function of particle size. Travel distance for suspended load (wash load plus some sands) ranges from 2 to 20 km/year whereas bedload consisting of pebbles and cobbles travel only 0.02 to 0.5 km/yr. For low gradient channels such as those found in portions of the Lake states and the southeastern US, residence times for sands can range from 50 to 100 years (Dissmeyer, 1976; Trimble, 1981; Phillips, 1993). Studies in the western U.S. show sediment storage times within active stream channels ranging from 5 to 100's of years depending on particle size and the type of sediment deposit (Megahan et al, 1980; Madej and Ozaki, 1996; Ziemer et al., 1991).

Reliability of Findings – For the research studies, the findings are reliable for the streams studied. For other streams, land use history, field observations by a specialist, and changes in stream cross sections can be used to determine if streams are transporting stored sediments.

Ability to Extrapolate Findings – Each stream needs to be evaluated to determine if the streams are transporting stored sediments from prior land uses.

Secondary Linkages – Stored sediment from prior land uses may contain chemicals and metals that can endanger human health.

Ability to Address Issues – There are a variety of channel geomorphology classification and evaluation techniques available to evaluate sediment transport and to determine if sediment is being taken out of storage. Methods are available to help stabilize stream bed and bank erosion problems, but they are often costly and of limited success.

Research Needs – Research is limited in this area. Research needs include: (1) development of reliable methods to route sediment through channels, (2) determining the chemical and metals attached to sediments being taken out of storage, and (3) in refining methods for determining the proportion of present sediment yield is from present land management activities and from prior land uses.

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Chapter 13

Synthesis and Integration

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July 14 99

Introduction

Drinking water is essential to life and social wellbeing and is often considered a “priceless” resource. However, because of it’s relative abundance and the self-purifying nature of the hydrologic cycle, society rarely acts as though water had a value equal to life itself (National Water Commission 1973). Instead, large quantities of water are wasted or polluted and there are extremely different views of how this “priceless” resource is managed (Mann 1993). Some conceptualizes water policies as being the product of the needs of industrial, agricultural and municipal interests (i.e. the hydraulic society view). Others view water policy as a set of illogical and conflicting compromises that have evolved from disorganized, non-scientific, haphazard, and short-sited decisions (i.e. the Cadillac Desert view). Still others view U.S. water policy as the quintessential example of a distributive political process (i.e. the pork-barrel view) or think that policies should be based on free-market principals (i.e. the user-pay view).

Regardless of how U.S. water policy has developed, the success in supplying water during the 20th century has created the public expectation that large quantities of high quality water should be supplied at minimum monetary and environmental costs (National Water Commission 1973). To meet the demands and expectations of the 21st century will require increasingly complex management (Reuss 1993). This chapter discusses the uncertainties associated with assessing municipal water supplies and reviews present research needs and future trends that are expected to affect the management of the resource.

Uncertainties in water supply assessment

As is clearly demonstrated in many sections of this book, the flow of water across the landscape, and its collection and distribution through a municipal water system is a complex and dynamic process. Because of the complexities and risks involved in supplying municipal water, some basic level of water treatment and quality monitoring are always necessary. However, it is equally certain that it is technically and economically impossible to monitor and treat for all contaminates, at all locations, at all times. The major objective in identifying and assessing potential contamination to source water is to insure that adequate levels of treatment will be undertaken so that a steady supply of safe water can be supplied at minimum costs (U.S. EPA 1997).

Water quality assessments and monitoring programs can be influenced by many technical and economic factors (MacDonald et al. 1991, Harmancioglu et al. 1992). Because the results of any assessment depend on how the data was collected, the limitations of the data need to be explicitly recognized when making decisions or management guidelines. Without full consideration of these uncertainties, managers increases the risk of making false decisions and may conclude that there are problems when there actually are none or decide there are no problems when they are present and form a significant threat.

In general, uncertainty and errors associated with water quality assessments can arise from; A) lack of sampling (i.e. not looking for a particular problem or contaminant); B) improper sampling (i.e. sampling at the wrong time or location or with the wrong equipment); and C) technical limitations (i.e. measurement and data management errors). Errors associated with calculating hydrologic budgets alone can be 20% or greater (Winter et al. 1981). Problems in collecting and managing automatically collected chemistry data from rivers can resulted in anomalies (i.e. missing days, incomplete data etc.) in more than one-third of the data (Bisbal and Ruff 1996). Likewise, "unaccounted-for" municipal water, or the difference between the amount of water produced by treatments plants and the amount legitimately consumed, are commonly 20 to 40 % of total production (World Bank 1993).

When considering the effectiveness of water supply assessments, it should also be recognized that not all environmental problems can be examined solely within the context of watersheds or topographic divides. Nor is the risk or consequences of inaccurate decisions evenly allocated across the landscape or within the public (Frederick 1993). Moreover, land use patterns, water supplies and demands, wastewater production and other environmental issues are not always constrained by either watershed or political boundaries. Furthermore, in many areas subsurface features can control groundwater water flow irrespective of surface topography and there can be large differences in the problems and techniques needed to manage upland and lowland areas within the same watershed.

Future Trends and Research Needs

The management of lands for municipal water supplies has been, and will continue to be, a major activity of the U.S. Forest Service. However, National Forests will continue to support other uses, including timber products, recreation, mining, fisheries, grazing, and conservation of biodiversity. In addition, relatively new uses like using National forest lands for carbon sequestration (De Lucia et al. 1999) or the recycling of wastewater (Sopper and Kardos 1973) will increase in the future. How these municipal water supplies will be managed in this multi-use environment will be affected by environmental change, technological change, and social and administrative considerations. In addition, effective management will require site specific knowledge of environmental processes and conditions.

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 parts of the globe during the 22nd century (Schlesinger 1997). Most global scale climate models predict a warmer, more humid world with increase fluxes in evaporation, precipitation and runoff in the next 50 years (Loaiciga et al. 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 long-term average hydrologic conditions and projected demands suggest that areas East of the Great Plains and in the Pacific Northwest will have water surpluses until 2040 (Gulidin 1989). In contrast, much of the Colorado river basin, the Rio Grande region, the Great Basin region, parts of California and the lower Mississippi currently have or will have water shortages by 2040. How historically based projections of supply and demand will vary with the predicted global-scale environmental change is currently a major research questions that is being addressed at regional levels (reference California and Southern studies).

In addition to global-scale environmental change, local and regional scale changes can be expected to have both direct and indirect influences on municipal water supplies. These changes can occur over years or decades and may include changes in the types or intensity of land uses within or adjacent to municipal watersheds, increases in air pollution and atmospheric inputs, or changes in the demand for different water uses (i.e. irrigation, recreation etc.). Changes in the successional status of forest, or the presence of exotic or noxious weeds are additional types of local changes that can affect the quantity and quality of source waters (*see chapters 3 and 4, Succession and Vegetation Management and*

Juniper example). Likewise, changes in existing management practices, like the tendency to harvest on steeper slopes (Stednick Chapter 4), or changes in the chemicals used in fisheries and vegetation management (Dolloff Chapter 6, Michael Chapter 4) can also influence environmental conditions that affect municipal water supplies.

In response to the importance of regional scales changes, ecological risk assessments are now being developed to evaluate the regional risks of specific environmental hazards (Graham et al. 1991). A generic problem that is often encountered when doing these assessments is the lack of region or ecosystem specific information and spatial data. Fortunately, technological advances in data acquisition and management are rapidly improving this situation.

Technological change

The ability to provide safe drinking water depends on the technology that is used to: A) measure the quantity of water and contaminants; B) run treatment plants and distribution systems; C) charge consumers for the resources they use. Future advances in the technology used to accomplish these tasks will also effect future municipal water management. In addition, technology and anthropogentic activities can be expected to create new chemicals, new pathogens and presently unknown water quality problems.

Increased competition for water resources is expected to increase the scrutiny of the methods used to manage water supplies. Fortunately, recent advances in the technology used to acquire hydrologic data are greatly improving our ability to measure and monitor water resources. For example, the monitoring of nutrient and pollution concentrations, and erosion and deposition in stream channels can be automated (Lawler 1991). Recent advances in remote sensing have also greatly increased the ability to estimate precipitation and evapotranspiration over large areas and the development of inexpensive field recorders and wireless communication systems allows faster and more extensive sampling of many hydrological processes. This type of "real-time" monitoring is currently being used to manage irrigation systems, water supply reservoirs, flood warning systems, and fish migrations in rivers.

While automated data collection techniques are essential to accurately monitoring hydrologic processes, without proper analysis the collection of massive amount of data can hinder rather than assist management. A major challenge for watershed management in the 21st century will be the development of analytical methods and institutional structures that can rapidly synthesizes environmental information so managers can make informed, defensible decisions in a timely fashion. In response to this challenge the Forest Service has begun to adopt guidelines for managing information (Spencer 1996) and computerized environmental decision support systems are being developed by a range of research organizations (Lovejoy et al. 1997). These decision support system are expected to eventually integrate real time hydrologic measurements with geographical information systems and multi-objective decision models. Multi-objective 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 which scientific, environmental, social information can be organized and synthesized (Hipel 1992). Nevertheless, their adoption from planning tools to real-time management tools will be aconsiderable challenge. Moreover, just collecting real time water quality information that will eventually be used in complex models can be a daunting task (Bistal and Ruff 1996).

The use of management guidelines and operating rules that are based on the life histories of organisms will also present new opportunities and challenges for managers of municipal water systems. Recent examples of this type of management include: the timing of stream water withdrawals and releases to minimize impacts on the migration of aquatic organisms (Bisbal and Ruff 1996, March et al. 1998, Benstead et al. 1999); the timing of fertilizer application to growth phases of agricultural crops to minimize nutrient runoff (Matson et al 1998); timing the abundance of grazing with the growing season of range and riparian vegetation (Buckhouse chapter 4); and the scheduling insecticide application to the life cycles of pests (Balogh and Walker 1992. Michael Chapter 4). 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.

Administrative change

Because of the high engineering and environmental costs associated with developing new water supplies, the emphasis in water management is shifting away from the development of new sources and toward the efficient and equitable use of existing supplies (Kneese 1993, Frederick 1993). In response to this shift toward more efficient use of existing resources, 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, storm water utilities, waste management districts, and watershed restoration groups that have recently been established (Mann 1993, Shabman 1993, Taff and Senjem 1996, Watershed Protection Techniques 1997). 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 or government subsidies for water and pollution control. This in turn generates inadequate operating funds and a further deterioration of services by the overextended agencies. To successful deal with these problems economists have championed "user pays" and “market based" approaches to water and pollution management (Busby 1955, Kneese 1993). For municipal water system this approach has recently meant changing rate structures to promote conservation or the privatization of water management services. Several different market-based approaches have been promote 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 inter-basin or inter-region water or pollutant transfers (Mann 1993, Taff and Senjem 1996).

While these market based approaches have been widely promoted by economists, non-economists and water resource managers have been less enthusiastic about their adoption (Brookshire et al. 1992, Taff and Senjem 1996). The principal stumbling blocks are usually the technological and organizational needs that are required to monitoring and enforce the complex and dynamic trade of water resources and pollutant effluent. Nevertheless, because recently developed data acquisition technology allows that pollution discharges and water withdrawals be continuous monitored it is generally considered that these market-based, polluter-based approaches will increase in the near future. However, regardless of the advances of data acquisition technology and decision based models, these administrative and technological changes will only be successful when based on site-specific information.

Site specific considerations

The social, ecosystem, and watershed level response to both global scale and local scales changes that can effect municipal watersheds are expected to be complex because different ecosystems and processes respond at different rates and in different magnitudes (Schimel et al. 1996, Schlesinger 1997). 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, road removal operations require site specific analysis since in some situations the disturbance caused by closure may accelerate rather than reduce erosion (Elliot Chapter 11). Likewise, the impacts of grazing also require understanding of the specific grazers and the local seasonal cycle of range land vegetation (Buckhouse Chapter 5). The response of atmospheric deposition of nitrogen also depends on the level of nitrogen saturation of the receiving ecosystem and the seasonal variations in plant growth and nitrogen use (Fenn et al. 1998, Verry Chapter 4). The abundance and frequency of herbicide use also depends on site conditions and can range from never to once every 20 to 150 years (Michael, Chapter 5). Furthermore, the width and composition of buffer zones needed to contain chemicals, reduce impacts of grazers, or maintain aquatic habitat can also be expected to be ecosystem and problem specific (Michaels and Buckhouse Chapter 5, Dolloff etc.). Regional risk assessments of water quality changes due to ozone-induced stress on coniferous trees also indicate that changes in the water quality are closely related to location-specific forest cover changes (Graham et al. 1991). Likewise, the impacts of accidental chemical spills or other historical legacies that alter water quality are site specific and often 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 papers in this volume. For example, the amount and impacts of recreation on site and water quality are regional, seasonal and episodic (Chapter 6). Likewise, 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 (Chapter 7). Grazing behavior and the potential impacts of grazing in water quality can also vary with season, species, and the age of individuals (Buckhouse, Chapter 5). While it is recognized that there are temporal differences in the potential of wildlife to contaminate water supplies and threaten human health, the life histories or wildlife species that are reservoirs of waterborne human pathogens are poorly understood (Chapters 5, 6, 7). Determining the timing and pathways of these pathogens is essential for evaluating the temporal and spatial variation of risk and remains a major research need.

In addition to differences in the level of use activity, the length of time a particular activity can effect water quality also varies with land use and between different ecosystems. Sediment yields or concentrations following timber harvesting typically decrease as a negative exponential relationship while nutrient pluses are short-lived (Stednick Chapter 5). Sediment yields from roads typically peak in first few years but can remain elevated for decades (Elliot Chapter 10, Swank Chapter 4). Likewise, the residence time of fecal contamination in streams can be on the order of weeks to months (Buckhouse, Chapter 5) while contamination from road-side fuel spills can last for years. Mining debris can cause acidification of surface and subsurface water for decades or longer (Wireman, Chapter 9). Developing methods and monitoring protocol to determine and predict critical time periods when water supplies have an increased risk of contamination is an additional challenge that will require the interaction of scientific information, technology, and administrative structures. The future success of these interactions will also depends on the availability of high-quality, spatially explicit, long-term environmental data. In many regions, the water quality assessments of municipal water supplies that are currently being undertaken will provide invaluable base line information for future managers.

Conclusion

Supplying municipal drinking water has been and will continue to be a major activity Forest Service lands. However, with increased demands on FS lands this management will become increasingly complex. To accomplish this, continued research and long-term monitoring will be needed to develop and maintain the scientific foundation needed to determine what is the acceptable amount of wildlife, visitors, harvesting, grazing, in-holdings or other uses can a drinking water supply tolerate. These levels will not only depend on the type of use and the ecosystem used, but also the economics and technology associated with water treatment and contaminant removal. The dramatic improvements in the water quality of U.S. that have occurred during the last few decades clearly demonstrates the success that integrated, continued management can have (Dzurik 1990). It is hoped that this volume will assist water resource managers to successfully identify critical problems and "Protect the best and restore the rest".

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Chapter 14

Examples of Watershed Assessments

Four examples of watershed assessments are presented in this chapter. Two are municipal watersheds, one is cumulative impacts down a stream, and the last is a green algae bloom resulting in a toxic impact. Each employed a different approach to assessing watersheds for impacts on drinking water quality.

City of Baltimore-Municipal Reservoirs, Incorporating Forest Management Principles and Practices

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Issues - 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. They serve as the water supply for over 1.5 million area residents. The reservoirs are surrounded by 17,580 acres of city owned forested land which was acquired between 1880 and 1955, to insure control of land use in critical areas immediately adjacent to the reservoirs. Forest management on the reservoir lands 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 for the recognized additional values in forested watershed lands. Along with reservoir protection, the harvest of forest products yielded revenue for watershed enhancements, and provided lumber Department of Public Works uses within the city.

In recent years, the reservoir lands have also come to be regarded as important sites for the conservation of regional biodiversity and dispersed outdoor recreation. While timber harvesting occurring on these lands initially fueled the concerns of private citizens and conservation organizations in 1989, uncontrolled access, and a rapid increase in recreational use also served as the impetus for the City to re-evaluate its management practices. At the same time the public agencies responsible for Maryland's Source Water Protection Assessment (Safe Water Drinking Act Amendments, 1996), focused on the protection of drinking water quality and availability, were expressing concern over the eutrophic nature of the three reservoirs, and their loss of storage capacity due to sedimentation.

The watersheds which serve as the primary source of water supply for the reservoirs are located in Baltimore, Harford and Carroll Counties in Maryland, as well as York County, Pennsylvania. The City-owned lands make up only a small fraction - on average only 7% - of the total area of the watersheds draining into each reservoir. These source water drainages are part of the urbanizing, and expanding, Baltimore - Washington metro area (4th largest in the US), though the Prettyboy and Liberty basins are still rural in character with an agricultural land use predominant. Preserving the quality of the water that flows into the reservoirs requires careful control of sediment, as well as point and non-point source pollutants.

A watershed management strategy is now being seen by all local, state and federal agencies as vitally important to the continued, efficient and economic provision of safe drinking water for the region's residents. The role of forests and of forest management, through its potential influence on water quantity, and ability to effect quality by filtering and sequestering various forms of soluble and solid pollutants coming from adjacent land uses, is recognized as a key component of the management of these drainages. Private forest landowners, through the application of forest conservation principles on their land, including the restoration of forest wetlands and riparian forests, and implementation of silvicultural practices to maintain a vigorous forest condition, are collectively enhancing the water quality of receiving streams and ultimately the reservoirs.

Ability to Address Issues - The Maryland Department of Natural Resources, 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. Forest Service, and the use of its NED-1 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, the water quality of first and second order streams. A separate recreational use survey will be conducted through contract with a regional university.

Goals for conservation were set through a series of twenty public meetings conducted by the City of Baltimore's Dept. 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; and
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 - Reservoir Technical Committee, and the Gunpowder Watershed Project, an 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 is being supplied through the Maryland Department of the Environment's Source Water Assessment Program, required by the 1996 Amendments to the Safe Drinking Water Act (Section 1453), and on the Liberty Reservoir drainage basin 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 within the Prettyboy and Loch Raven drainage basin, 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 groundwater contamination. These reports are also indicating that nutrient contamination is widespread throughout the Prettyboy and Liberty drainage basins as well, with current levels at the within the same range found at the Loch Raven basin.

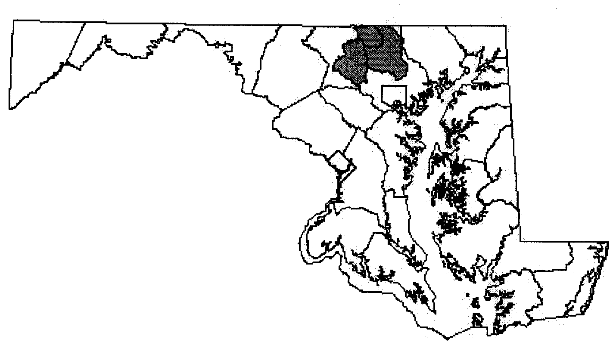
Source water protection strategies, being developed by these groups and associated state agencies, highlight the need to conserve the existing forest land base in a healthy and vigorous condition, while restoring forest wetlands and riparian forests for their functional ability to filter sediment and other suspended solids, sequester pollutants within 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 groundwater 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 Maryland Department of Natural Resources has developed a methodology that locates and ranks potential restoration sites. Potential forest wetland restoration sites are located by identifying hydric soils, which lack natural vegetative cover. This system locates potential riparian forest restoration sites by identifying stream segments that lack forest cover and assesses their potential to improve water quality. The potential is assessed by assigning a weighed rank based upon the nutrient loading potential of adjacent land uses, whether the ownership parcel is greater than 10 acres, and stream order (lower order streams receive higher ranking).

Interest in the management of the City owned, and surrounding, forest lands is keen and public support critical to the plan's successful implementation. The Friends of the Watershed, an existing city sponsored citizen's advisory group, will be invited to initially review data sets being collected and 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 by multiple scales (unit to ecoregion) and time frames, the city owned lands will be evaluated to determine their potential to serve as buffers to adjacent land uses, support an increasing desire on behalf of the growing urban population for outdoor recreation, while at the same time participating 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 their properties for the multiple values consist with its stated goals.

Finally, the plan will offer forest management guidance to local, state and federal agencies concerned with the continued decline in the region's forest land base, which has been compounded by the cumulative impacts of pollution, fragmentation and habitat loss. It will support and clarify the function of forest resources as integral to the long-term sustainability of local watersheds and lead to the incorporation of forest management techniques in watershed strategies concerned with water quality.



Hebgen Lake Toxic Algae Bloom of 1977

The moral of this story is to take reasonable precautions to protect human lives and pets in case a blue-green algae bloom in a heavily used recreation reservoir or lake suddenly becomes toxic.

In late June of 1977, I got a call from Claude Coffin, recreation staff officer on the Hebgen Lake Ranger District at West Yellowstone, Montana. 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. A long tenured employee there, Claude had never seen anything like this before, and asked me to hurry on down to West Yellowstone and try to determine what was happening.

At the time, I was the hydrologist for the Gallatin National Forest, which includes that area. Before leaving for West Yellowstone, I called the Gallatin county sanitarian and told him about what I had heard and asked if he wanted to go with me. He did, and grabbing his sampliing gear and some "Area Closed" signs, we were off. As we drove the 83 miles through the beautiful Gallatin River canyon and passed the Big Sky resort, we discussed the possibilities and what we needed to do upon arrival at the lake.

We arrived at the campground and 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 some dead cattle lying near the lakeshore beyond the campgound fence. We could also see a green scum on the surface of the water that was different from the algae we had seen there in previous years - this coating resembled thick green pea soup, was odorless, and went at least 50 feet offshore. We took water samples with our sampling gear, including the green algae in the samples. We put them into iced chests, and decided they should go to Montana Water Quality Bureau in Helena for analysis. The sanitarian posted his "Area Closed" signs at the campground and it was closed down that day.

The next day we took the samples to Helena and met with Water Quality Bureau scientists. After they heard what we had found, they made arrangements to go to West Yellowstone after phoning some toxic algae experts (Carmichael, Juday and Keller) and reporting this episode. These men arrived a few days later and began intensive studies of the algal bloom. They identified the culprit as *Anabena flos-aquae*, a blue-green algae that sometimes develops a toxin (anatoxin-a) which is released into the water. With the 4th of July holiday approaching, a meeting was held to decide what protective measures needed to be implemented to prevent any more loss of pets, 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 non-toxic 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 foot 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 which produce a naturally high concentration of nutrients. As a result, phosphate is relatively high. The reservoir is nitrogen limited. Juday et.al. (1981) classified the main part of Hebgen Lake reservoir as mesotrophic and the Grayling Arm as eutrophic. They also found the Anabena flos-aquae algae disappeared about one kilometer out in the main part of the lake. Apparently the chemistry of that water was inhospitable to the *Anaben*a.

This case study is extracted from R.E. Juday, E.J. Keller, A. Horpestad, L.L. Bahls, and S.P. Glasser. 1981. A Toxic Bloom of Anabena Flos-Aquae in Hebgen Lake Reservoir, Montana, in 1977. In: W.W. Carmichael ed. The Water Environment: Algal Toxins and Health. Plenum Press, New York. 491 pp.

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Managing the Shift from Water Yield to Water Quality
on Boston's Water Supply Watersheds

Thom Kyker-Snowman, Natural Resources Specialist
Massachusetts Metropolitan District Commission, Division of Watershed Management

Boston's drinking water derives from surface reservoirs within three major watersheds - the Quabbin, Ware River, and Wachusett (see map). These watersheds total in excess of 225,000 acres, about 40% of which 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 daily to accommodate the drinking water demands of 2.5 million people, about 40% of the population of the Commonwealth. Water from this system is currently treated (chlorine and chloramines for disinfection; fluoride to promote healthy teeth; soda ash and carbon dioxide to prevent corrosion of pipes), but not filtered. The desire to avoid the costs as well as many other ramifications of filtration is at the center of current watershed management decision-making for this system. This represents a dramatic shift in objective from the focus on water *quantity,* which has dominated the bulk of the history of Boston's water suppliers.

Since the settlement of Boston in the 1600's, its citizen's have looked ever farther west to meet the steady demand for more water. In 1795, the "Aqueduct Corporation" was created to tap Jamaica Pond in Roxbury to supply the 20,000 Boston inhabitants at that time. In 1848, Lake Cochituate was added, and in short order from 1870 to 1880, the Sudbury and Framingham Reservoirs came on line. By 1895, Boston's population exceeded 500,000 and the metropolitan area exceeding a million. The Wachusett Reservoir, the largest reservoir in the world at the time, was built by 1908 and added a 65 billion gallon capacity to the system. Still not enough to keep up with the growing demand, Boston next tapped the Ware River with an aqueduct to the Wachusett Reservoir, and finally constructed the 412 billion gallon Quabbin Reservoir within the valley of the Swift River, disincorporating four towns in the process, while providing Depression-era labor for thousands of underemployed workers.

Despite these efforts, water quantity persisted as a concern. In 1967, just twenty years after Quabbin filled to capacity, a severe drought brought the Reservoir to 45% of its capacity and brought skeptics to worry that it would never fill again. While Quabbin filled to capacity again by 1976, water demands were exceeding the safe yield from the system (300 million gallons/day) by almost 50 million gallons 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 yield from its land holdings, primarily by clearcutting 2,000 acres of red pine plantations and converting these to fields, a practice which was estimated to provide an additional 3-400 million gallons of water annually. After a 1989 drought dropped Quabbin to a 17 year low, authorities declared a water emergency. Water conservation efforts (spurred in part by rising rates that accompanied a clean up of Boston Harbor from ratepayer revenues), and an aggressive leak detection and repair program have brought water

consumption dramatically lower, so that today the daily draw on the system is 50 million gallons below its safe yield.
In addition to initial efforts to reduce forest cover (to increase yield) by converting pine plantations to grass, MDC-DWM postponed management of an inflated deer population because it was felt that the impact of deer on the understory included an increase in water yield. The deer population at Quabbin had grown to nearly 6-8 times the statewide average of 8-10 deer per square mile, under the hunting restrictions that were standard watershed practices for lands brought under MDC-DWM control. Early forest management plans had acknowledged the impact of this population on the understory, but the emphasis on water yield made it easier to chose to avoid the difficult politics associated with starting a deer management program, especially following fifty years of hunting prohibition.

Changes in drinking water laws and regulations have brought about dramatic changes in the approach to managing natural resources on the watershed lands surrounding MDC-DWM's unfiltered surface supplies. The federal Safe Drinking Water Act became law in 1974, and set national standards for maximum contaminant levels and treatment techniques. Amendments to the SDWA in 1986 established a priority for using filtration as a dominant treatment technique. EPA addressed this priority through the Surface Water Treatment Rule of 1989, 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. MDC-DWM currently owns and controls 64% of the Quabbin watershed, 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 this land and pay tax substitutes to the local towns would persist. Similarly, recreation is carefully limited on these watershed lands, but it would be increasingly difficult to resist these pressures in the absence of a requirement for natural filtration of Boston's drinking water.

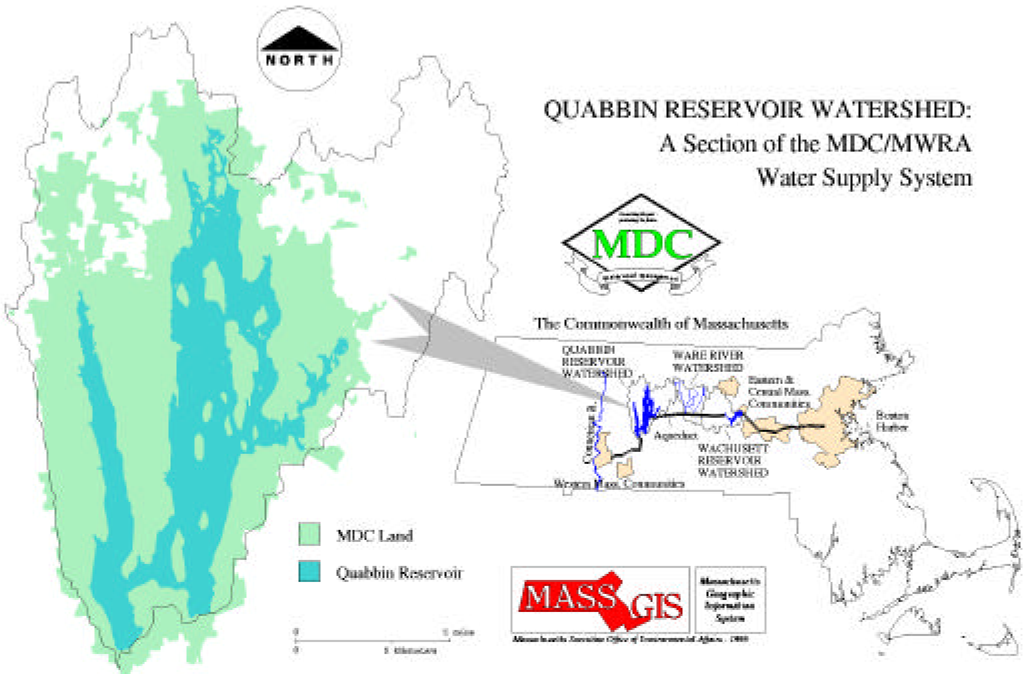
The combination of reduced pressure to increase yields and a strengthening desire to avoid filtration shifted the management focus at Quabbin away from water production and sharply toward water quality protection. From the natural resources perspective, this meant demonstrating that wildlife and forest cover were being managed to avoid degradation and, if possible, to provide improvements in the natural filtration process. Two major wildlife issues were met squarely along these lines: waterfowl (in particular gulls and geese) and white-tailed deer. Seagulls have threatened the maintenance of water quality standards when they spend their days wandering landfills and move by the thousands to roost on open surface water supplies, transporting pathogens that can threaten human health. MDC-DWM has devised an elaborate "gull harassment" program that deals with the problem by moving roosting birds far from the reservoir outlets.

The threat posed by high populations of white-tailed deer is directly related to the prevention of regeneration of forest cover lost to natural or human disturbance. This problem is exacerbated at Quabbin by the threat of major overstory losses associated with catastrophic hurricanes that recur in New England every 100-150 years, the most recent of which 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. Application of this model predicted in 1992 that 75% of the conifers and 50-75% of the hardwoods on 37,000 acres, plus 50-75% of the conifers and 25-50% of the hardwoods on 13,000 acres at Quabbin 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. MDC-DWM engaged in a lengthy, multi-year 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. This would then constitute 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 fifty 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 (for instance, tree stems above 4.5 feet in height averaged 170 per acre prior to the hunt but are currently approaching 1,000 per acre; and wildflowers like trillium and marsh marigold have reappeared after a long absence).

In addition to recovering the watershed's regenerative capacity, plans called for deliberately diversifying both the age and the species structure of the watershed forest cover. In its simplest form, this objective calls for maintaining 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 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 of 3-6 million board feet annually, 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 lots.

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. 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 412 billion gallon 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.


NORTH
QUABBIN RESERVOIR WATERSHED:
A Section of the MDC/MWRA
Water Supply System
MDC
The Commonwealth of Massachusetts
QUABBIN
RESERVOIR
WATERSHED
WARE RIVER
WATERSHED
Eastern &
Central Mass.
Communities
Boston
Harbor
Aqueduct
WACHUSETT
RESERVOIR
WATERSHED
Western Mass. Communities
MDC Land
Quabbin Reservoir
MASS GIS
Massachusetts
Geographic
Information
System

Cumulative Impacts of Landuse on Water Quality
In a Southern Appalachian Watershed

Wayne T. Swank

(This example is excerpted from a paper by Bolstad and Swank,1997)

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 landuses. Variables sampled include pH, HCO_3^{2-}, conductivity, NO_3-N, NH_4+-N, PO_4^{3-}-P, Cl^-, Na^+, K^+, Ca^{2+}, Mg^{2+}, SO_4^{2-}, SiO_2, turbidity, temperature, dissolved oxygen, total and fecal coliform, and fecal streptococcus. Landcover/landuse was interpreted from 1:20,000 aerial photographs and entered in a GIS, along with information 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.

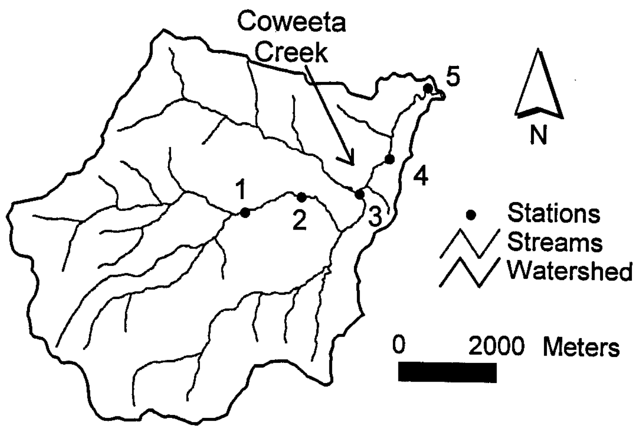


Figure 1. Watershed boundary and stream sampling locations in the Coweeta Creek Watershed in Western North Carolina. Stations 1 through 5 are arranged down the stream gradient on Coweeta Creek. First order streams are not shown.

Five water quality monitoring stations were located over 8.7 km of Coweeta Creek (Figure 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 (Table 1). Sites were selected to encompass incremental additions and a variety of landuses. 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 landuse features such as residences along the stream, grazing and other agricultural practices, plus additional roads. Stations 2 through 4 were characterized by a two to six-meter wide riparian shrub strip (chiefly *Alnus, Rubus,* and *Salix*) 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.

Table 1. Summary data for the catchments above five sampling stations.

Characteristics upstream of Sample Station	Upstream 1	Sampling Station Number 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 (#/100 ha)	0.37	3.06	5.36	6.01	9.23

Stream water 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.

Baseflow Water Quality

Water quality was good during baseflow conditions over the three-year study period. Concentrations of most solutes averaged less than 1 mg/l, 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 general low, typical for the southern Appalachians (Figure 2), averaging less than 6 NTU 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^{2-}, SO_4^{2-}, Cl-, conductivity, turbidity, and temperature general increased downstream from Station 1 to 5.

Mean baseflow levels for total coliform, fecal coliform, and streptococci counts increase from three to eight-fold downstream (Table 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 baseflow samples represent periods when there is little or no overland flow input from adjacent lands.

Table 2. Summary water quality data from baseflow grab samples at each of the five sampling stations. (Means)

	Station Number				
Variable	1	2	3	4	5
Total coliform	9470	13660	40040	30740	52140
Fecal Coliform	200	340	460	1130	840
Fecal streptococcus	710	1310	2180	1590	1840

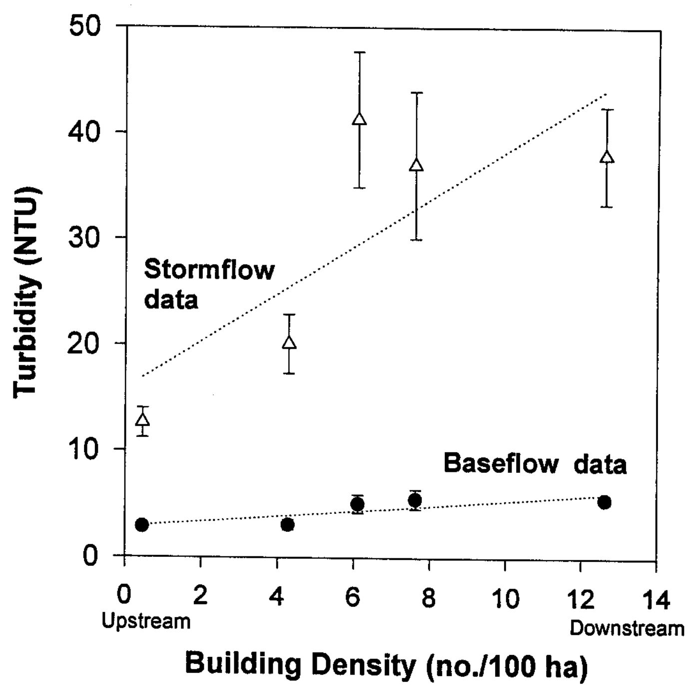
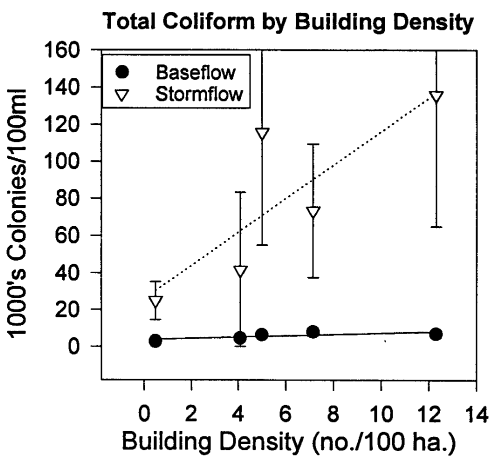


Figure 2. Mean and standard error (bars) for turbidity, plotted against building density for each sampling condition (baseflow and stormflow).

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 non-significant to quite large (turbidity, Figure 1).

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 two- to three-fold during storm events compare 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 (Table 3, Figure 3).



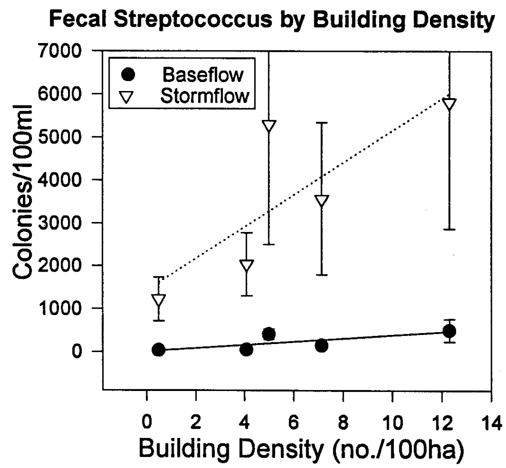


Figure 3. Mean and standard error (bars) for total coliform and fecal streptococcus, plotted against building density for each sampling condition (baseflow and stormflow samples). (Building density increase down stream).

Table 3. Summary water quality data from stormflow samples at each of the five sampling stations. (Means)

	Station Number				
Variable	1	2	3	4	5
Total coliform	18790	34640	_	77160	98390
Fecal Coliform	880	130	_	970	1260
Fecal streptococcus	450	8710	_	3260	4190

Conclusions

In summary, this work identifies consistent, cumulative downstream changes in Coweeta Creek concomitant with downstream changes in landuse. 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.

References:

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