Part III:

Effects of Vegetation Management on Water Quality

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

Chapter 10

Timber Management

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Introduction

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

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

Erosion/Sedimentation

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

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

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

Issues and Risks

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

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

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

Findings from Studies

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

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

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

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

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

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

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

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

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

continued

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

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

Conc. = concentration, WS = watershed.

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

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

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

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

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

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

Reliability and Limitations of Findings

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

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

Research Needs

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

3. Research is needed on routing eroded soil to streams. Erosion does not equal suspended sediment. Measured erosion rates do not or should not imply that eroded soil is reaching the stream channel. Suspended sediment monitoring is not difficult, but requires labor and equipment that may not be available. For source areas the question is: Do suspended sediment concentrations best measure the effects of site disturbance?
4. Recommendation and design of vegetative filter strips are often based on width only. Research needs to better define the characteristics that control sediment movement including slope, vegetative and litter cover, runoff velocity and volume, surface roughness, and microtopography of the filter strip and disturbed area above.
5. Research is needed on monitoring of stream channel geomorphologic features, which may provide a good measure of land-use effects, particularly multiple or cumulative effects. Increased annual water yield from timber harvesting has been well documented, but the effect of timber harvesting on peak flows is less clear. Can this altered hydrology increase sediment transport from in-channel sources and result in changes in channel morphology? Conversely, how much increased sediment input can a stream segment receive without changes in channel morphology?

Key Points

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

1. Careful planning, supervising, and implementing of forest practices.
2. Keeping the treatment area small and hydrologically isolated.
3. Leaving adequate filter strips between treatment areas and streams.
4. Maintaining ground cover in the treatment area to reduce surface runoff and erosion, and increasing the effectiveness of filter strips to trap eroded soil before it enters the stream.
5. Operating during the season with the lowest erosion risk.

Stream Temperature

Issues and Risks

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

Findings from Studies

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

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

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

Reliability and Limitations of Findings

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

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

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

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

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

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

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

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

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

Research Needs

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

Key Points

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

Nutrients

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

Issues and Risks

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

Findings from Studies

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

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

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

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

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

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

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

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

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

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

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

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

Reliability and Limitations of Findings

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

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

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

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

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

Research Need

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

Key Point

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

Fertilizer

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

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

Issues and Risks

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

Findings from Studies

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

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

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

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

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

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

NH_4-N = ammonium nitrogen; NO_3-N = nitrate nitrogen.

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

Reliability and Limitations of Findings

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

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

Research Needs

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

Key Points

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

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Chapter 11

Forest Succession

Wayne Swank[1]

Introduction

The effects of forest management activities on water quality are generally of the greatest magnitude in the first several years after disturbance. However, during long-term succession and regrowth of forest ecosystems, changes in physical, chemical, and biological parameters of streams may occur.

Nutrients

Issues and Risks

After a forest disturbance such as harvesting or fire, nutrient levels in streams may be elevated during early successional stages until the forest matures (see chapter 10). Nitrate concentrations can be elevated for a few to many years depending upon whether the watershed is nitrogen limited or saturated (see chapter 3 for discussion of nitrogen saturation).

Findings from Studies

Changes in stream inorganic chemistry and sediment yield were observed over a 20-year period after clearcutting by cable logging of a 146-acre [58-hectare (ha)] Southern Appalachian watershed (Swank and others, in press). Stream nutrient concentrations and fluxes showed small increases after harvest, and responses were largest the third year after treatment. Nitrate-nitrogen (NO_3^{-1}) was an exception. The initial increase in nitrate was from <0.1 milligrams (mg) per liter to 0.8 mg per liter (fig. 11.1) and increased net nitrogen export of 1.16 pounds per acre [1.3 kilograms (kg) per hectare] the third year after harvest. However, later in succession (15 to 20 years), nitrate concentrations exceeded values observed the first several years after clearcutting. This response is partially attributed to reduction in nitrate uptake due to vegetation mortality, changes in species composition, and nitrogen release from decomposition of woody plants.

Other long-term research in eastern forested watersheds (Edwards and Helvey 1991, Swank and Vose 1997) shows that as forests mature, less nitrogen is retained in the watershed and stream nitrate concentrations increase. These long-term studies support findings of shorter term stream chemistry surveys. A survey of streamwater chemistry in 57 watersheds along successional and elevational gradients was conducted in the White Mountains of New Hampshire (Vitousek 1977). Differences in successional status among watersheds were found to be important in controlling nitrate and potassium concentration. Streams draining old-aged forests had higher concentrations of nitrate, potassium, and other solutes than did streams draining intermediate-aged forests at the same elevation. Spruce-fir (*Picea* spp.-*Abies* spp.) watersheds with no record of logging had streamwater nitrate concentrations of about 3 mg per liter, while spruce-fir watersheds logged 30 years previously had nitrate concentrations <0.5 mg per liter.

Another survey of 38 streams draining partially or entirely clearcut watersheds was conducted in New England—(Martin and others 1985) on northern hardwood sites in New Hampshire, Maine, and Vermont; in central hardwood forests in Connecticut; and in coniferous forests in Maine and Vermont. Streams draining watersheds that had been partially or entirely clearcut in the previous 2 years were selected. There were no apparent changes in stream nutrient concentrations from many of the ecosystems, and the largest concentration increases were for nitrate, calcium, and potassium in northern hardwoods of New Hampshire. Inorganic nitrogen (nitrate plus ammonium) increased to an average of 2 mg per liter (Martin and others 1985). However, elevated solute concentrations appear to be short-lived, even in streams draining successional northern hardwood forests in New Hampshire (Hornbeck and others 1987). Moreover, early stream chemistry changes after clearcutting were considered insufficient to cause concern for public water supplies or for downstream nutrient loading (Martin and others 1985).

In the Pacific Northwest, forest-successional stage is not always a good predictor of nitrate concentration in streamwater. For example, at the H.J. Andrews Experimental Forest in Oregon, forest harvest increased annual nitrate

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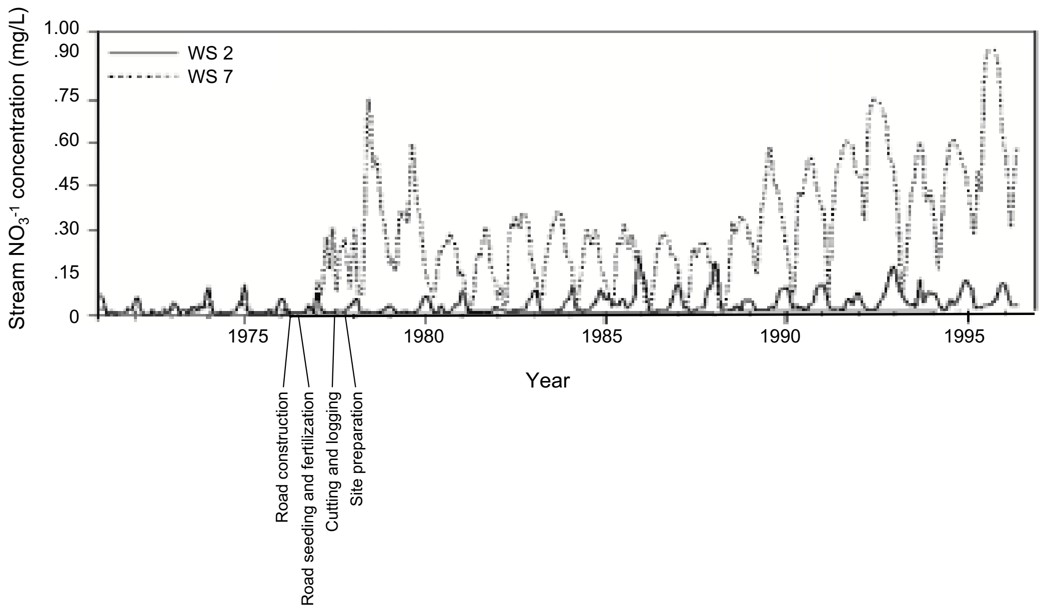

Figure 11.1—Mean monthly concentrations (flow weighted) of nitrate (NO_3^{-1}) in streamwater of a clearcut, cable-logged, hardwood-covered watershed (WS7) and an adjacent watershed (WS2) during calibration, treatment activities, and postharvest period, Coweeta Hydrologic Laboratory, North Carolina.

concentration from predisturbance levels of 0.001 mg per liter to 0.036 mg per liter (Martin and Harr 1989), but nitrate concentration returned to predisturbance levels within 6 years. Further, a 20-year postdisturbance record from a pair of treated and untreated watersheds at the experimental forest suggests that nitrate concentrations in streamwater remain very low in both watersheds once the clearcut watershed recovers from the immediate effects of disturbance.[2] At the H.J. Andrews Experimental Forest, the ecosystem is highly nitrogen-limited, and vegetation imprint on nitrogen fluxes may be overridden by rapid immobilization of any available nitrogen by soil microbiota.

An extensive synoptic water-quality assessment was conducted on numerous streams in the Great Smoky Mountain National Park in the Southern Appalachian Mountains (Flum and Nodvin 1995, Silsbee and Larson 1982). Concentrations of nitrate in streams draining watersheds that had been logged prior to park establishment were significantly lower (one-half) than the nitrate concentrations in unlogged watersheds at similar elevations.

The magnitude of stream nitrate concentrations associated with long-term forest succession depends on a number of factors, such as levels of atmospheric nitrogen deposition, the type and rapidity of forest regrowth, soil microbial activity, and soil physiochemical reactions. Stream nitrate levels rarely exceed 5 mg per liter and are below current drinking water standards. The nitrate, however, may contribute to stream acidification, particularly during spring snowmelt when nitrate concentrations peak in the Northeastern United States (Murdock and Stoddard 1992).

Reliability and Limitations of Findings

Existing evidence for changes in stream chemistry with forest succession is based upon well-established programs of long-term research and is quite reliable. However, findings are limited in scope to select forest ecosystems in the United States.

Limited evidence indicates that stream nitrate concentrations for older hardwood forests of the southern and central Appalachian regions are higher than for younger successional forests. However, site-specific research shows that nitrate levels can vary substantially even during early succession (first 20 years), although the general applicability of findings is unknown. Assessments of nitrate levels in streams draining successional forests in New England show mixed responses and appear to be ecosystem specific. Very limited information on stream nitrate is available for successional forests in the Pacific Northwest. Current findings

[2] Personal communication. 1999. Kristin Vanderbilt, Graduate Student. Oregon State University, Corvallis, OR.

indicate that elevated nitrate concentrations following clear-cutting are short-lived and return to predisturbance levels early in succession.

Research Needs

1. Long-term assessments of stream chemistry changes associated with forest succession are lacking for most major forest ecosystems in the United States. From a public drinking water perspective, synoptic stream nutrient surveys across a range of forest types and stand ages with known disturbance histories would greatly enhance planning information for managers.
2. There is a large knowledge gap in nutrient concentration changes associated with storm runoff events. Such information is most important where water supplies are derived from forested headwaters with rapid streamflow responses to precipitation, e.g., watersheds with shallow soils, steep slopes, intense rainfall, and rapid snowmelt.

Sediment

Issues and Risks

Stream sediment may also exhibit long-term dynamics after forest disturbance. Logging roads associated with harvesting activities are frequently the major source of sediment to streams and are a potential legacy to consider when evaluating sources of sediment in drinking water (see chapters 3, 9).

Findings from Studies

A synthesis of long-term sediment yield responses following forest watershed disturbances is provided by Bunte and MacDonald (1999). Based on studies in Oregon and New Hampshire, they identify three kinds of potential responses in postdisturbance sediment yields:

1. Sediment yields remain high for a number of years after disturbance due to a large sediment pulse to the stream from a storm or other disturbance. That is, sediment from up-stream storage areas or destabilized hillslopes and channels continues to be released;
2. Sediment yields decline below average annual yields after disturbance when sediment storage is depleted by a major sediment transport event; and
3. Sediment yields rapidly return to predisturbance conditions because excess material has moved through the system.

Recent findings in the Southern Appalachians provide an example of the first type of response where sediment yield remains high for a number of years during forest succession (Swank and others, in press). A cable-logged, clearcut watershed required only three contour access roads because logs could be yarded 1,000 feet (305 meters) with the cable system. Record storms (15 inches or 38 centimeters) in the last 2 weeks of May 1976, prior to grass establishment, eroded both unstable soil and hydroseeded materials from the roads. Roads were the source of elevated sediment yield as illustrated by soil loss measured at a gaging station in the stream immediately below a road crossing in the middle of the catchment (fig.11.2A). In those 2 weeks of May, sediment yield was nearly 55 tons [50 metric tonnes (Mg)] from 0.21 acre (0.085 ha) of road contributing area (road-bed, cut, and fill). In the ensuing period of road stabilization and minimum use (June to December 1976), soil loss was low but accelerated again briefly during the peak of logging activities (fig. 11.2A). In the next year, soil loss below the road declined to baseline levels.

The pattern of sediment yield at the base of the second-order stream (fig. 11.2B, gaging site) draining the watershed was different from the pattern of sediment loss from the roads. Following an initial pulse of sediment export from the watershed, sediment yield remained substantially elevated during and after logging. In the 3-year period between 1977–80, the cumulative increase in sediment yield was 240 tons (218 Mg) (fig. 11.2B). During the next 10 years, sediment yield declined with a cumulative increase in export of 240 tons (218 Mg). The rate of sediment yield over the 5- to 15-year period after disturbance was about 300 lb per acre per year (336 kg per hectare per year), or 50 percent above pretreatment levels. The long-term sediment yield data illustrate a lag or delay between pulsed sediment inputs to a stream and the routing of sediments through the stream channels. In the absence of significant additional sources of sediment to streams on the watershed, annual sediment yield at the base of the watershed was still substantially above predisturbance levels at least 15 years later. Thus, there appears to be a continual release of sediment from upstream storage that was primarily deposited from road crossings of streams during exceptionally severe storms.

Reliability and Limitations of Findings

Few studies have documented the long-term effects of management practices on sediment yield. As pointed out in chapter 10, increases in sediment yields from timber management activities are typically considered to be short-lived. However, unique conditions during management can lead to elevated stream sediment later in forest succession. The importance of this process is site-specific and requires

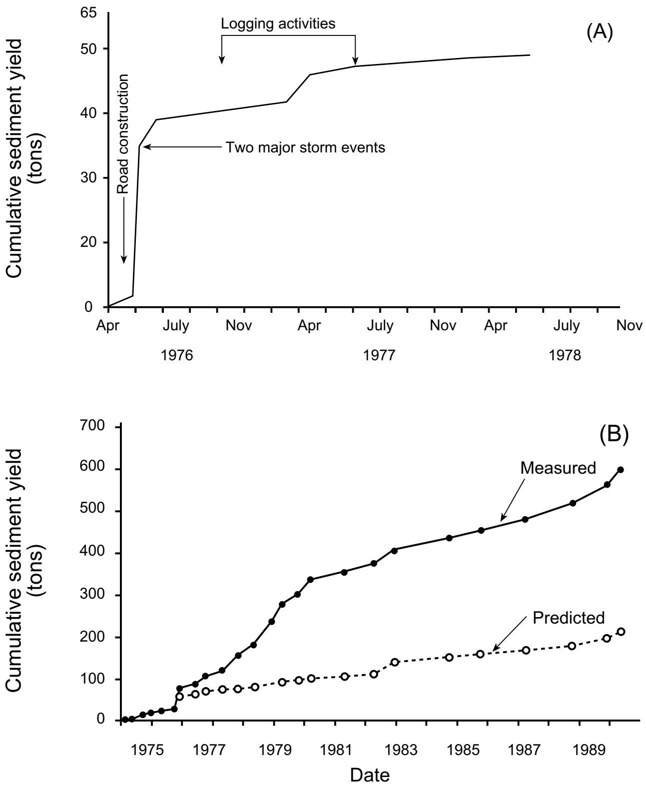

Figure 11.2—Cumulative sediment yield measured on a clearcut, cable-logged, hardwood-covered watershed: (A) in one of the first-order streams below a logging road during the first 32 months after treatment and (B) in the ponding basin of the second-order stream at the gaging site during 15 years after treatment. Predicted values are based on pretreatment calibration of sediment yield with an adjacent control watershed, Coweeta Hydrologic Laboratory, North Carolina.

that each stream be evaluated to assess the legacy of past management practices on current levels of stream sedimentation.

Research Need

Recommendations for future research related to this topic are given in chapter 10.

Key Points

In the long term, forest harvesting practices alone may have little deleterious impact on stream sediment and chemistry, which are of primary concern in drinking water. However, other past and present land uses affect present sediment and nitrate concentrations in streams. Sediment and nitrate yields associated with early successional development of forest may be in addition to yields from other past and present land uses. It is important to consider successional impacts along with the cumulative impacts of other past and present land uses across the landscape when assessing impacts of land management on drinking water sources.

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Chapter 12

Fire Management

Johanna D. Landsberg and Arthur R. Tiedemann[1]

Introduction

The effect of wildfire on drinking water was graphically demonstrated when the Buffalo Creek fire in Colorado in 1996 was followed by heavy rains, forcing municipal water supplies to shut off, one of Denver's water treatment plants to close, months to be spent cleaning a water-supply reservoir, and the Coors Brewing Company to bring in water by truck (Illg and Illg 1997).

Fire, both wild and prescribed, has the potential to alter physical, chemical, and biological properties of surface water that originates from burned wildland areas. Nonpoint-source pollution from wildland after fire can impair the suitability of water for drinking and other purposes. New plans for widespread use of prescribed fire to solve forest health problems create an urgent need to fully understand the water-quality consequences of increasing the occurrence of fire. Fire management activities (like retardant application, fireline construction, and postfire rehabilitation) also have potential effects on water quality.

The most important effects of fire on drinking water source quality include sediment and turbidity or both, water temperature, and increased nutrients in streamflow. In this chapter, we review results of research on the response of the above water-quality variables to fire, fire management activities, and fire rehabilitation measures. Much of the information comes from reports on wildfires. We would expect the magnitude of streamwater-quality changes after prescribed fire to be less than those observed after wildfires and some broadcast slash burns. It is unlikely that prescribed fire would consume as much forest floor and understory, or kill as much overstory, as would a wildfire because prescribed fires are usually conducted under conditions deliberately chosen to produce burns of low severity.

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Sediment and Turbidity

Issues and Risks

Suspended sediment is the major nonpoint-source pollution problem in forests (Society of American Foresters 1995). Beschta (1990) reported that sediment and turbidity are the most significant water-quality responses associated with fire. Turbidity has no direct health effects but can interfere with disinfection and provides a medium for microbial growth. Thus, it may indicate the presence of microbes (U.S. EPA 1999). See chapter 2 for more discussion on the effects of sediment on drinking water.

Findings from Studies

To understand research findings about sediment production and its impacts, one must be familiar with the units of measurement in which sediment is reported. Suspended sediment is particles carried in suspension and is measured by filtering and drying a known volume of water. Suspended sediment is expressed in parts per million (ppm), or as turbidity in nephelometric turbidity units (NTU's), which is a measure of the cloudiness of the water. These methods measure different characteristics of water, and it is difficult to correlate the results of one method with results of the other. The standard turbidity method (U.S. EPA 1999) uses NTU's. We found only two studies of fire effects that reported results in NTU's [equivalent to Jackson turbidity units (JTU's)] from American Public Health Association (1976) (table 12.1); all others reported sediment in parts per million (table 12.2). Beschta (1980) found that a relationship between suspended sediment and turbidity can be established but that the relationship differs significantly among watersheds. He suggested that the relationship must be established on a watershed-by-watershed basis. Recognizing this difficulty, Helvey and others (1985) determined the relationship between sediment in parts per million and turbidity in NTU's for three catchments in northcentral Washington and found the relationship to be strong (Helvey and others 1985). With this strong relationship and the equations developed, sediment measurements, in parts per million, can be converted to turbidity measurements, in

NTU's (fig. 12.1). This relationship has not been tested in other geographic areas or plant community types, so caution is advised when applying it beyond its original limits. It does point out the need to use a standard method or to establish relations between suspended sediment and turbidity for each watershed or stream system in question.

Our interest here is on the effects of fire on sediment measured in NTU's. Wright and others (1976, 1982) found that slope plays an important role in the amount of turbidity in streamflow after broadcast burning oak-juniper (*Quercus* spp.-*Juniperus* spp.) watersheds in central Texas. Turbidity changes (table 12.1) after burning were most pronounced in the steepest watersheds, with levels reaching 230 JTU's.

Studies of suspended sediment (table 12.2) show that the range of the prefire or control values is 1 to 26 ppm. Values obtained after fires indicate that fire has a profound effect on sediment movement.

Table 12.1—Water turbidity, in Jackson turbidity units (equivalent to nephelometric turbidity units), after fire alone or in combination with other treatments

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - Jackson turbidity units - -		
Prescribed fire, pile, and burn	Juniper	Central Texas			Wright and others 1976
		3 to 4% slope	12	12	
		8 to 20% slope	20	53	
		37 to 61% slope	12	132	
Pile and burn	Juniper	Central Texas	12	162	Wright and others 1982
Pile, burn, and seed			12	72	

Table 12.2—Suspended sediment concentration in streamflow after fire alone or in combination with other treatments

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - Parts per million - - - - -		
Wildfire	Taiga	Interior Alaska	3.7 –10.6	2.6 – 6.0	Lotspeich and others 1970
Clearcut, slash broadcast burned	Douglas-fir	Western Oregon	2	56 –150	Fredriksen 1971
Wildfire	Ponderosa pine, Douglas-fir	Eastern Washington	Not known	1,200[a]	Helvey 1980
Pile, burn	Juniper	Central Texas	1.1	3.7	Wright and others 1982
Pile, burn, and seed			1.0	3.7	
Prescribed fire	Loblolly pine plantation	Upper Piedmont, South Carolina	26	33	Douglass and Van Lear 1983
Wildfire	Lodgepole pine, Douglas-fir, ponderosa pine, western larch	Glacier National Park, MT	< 3	15 – 32	Hauer and Spencer 1998

[a] Maximum value attained.

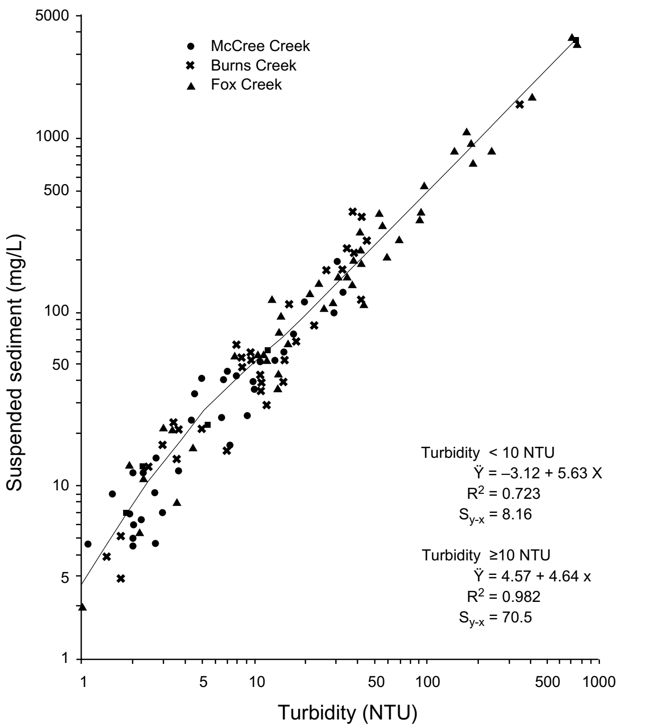

Figure 12.1—Relationship between turbidity in nephelometric turbidity units (NTU) and suspended sediment parts per million (ppm) (Helvey and others 1985).

Sediment yield has been measured in pounds per acre per year in many studies because of the concern for soil loss after fire. Sediment yield varies widely as a consequence of fire or forest harvest and fire (table 12.3). This variability reflects numerous interacting factors: geology, soil, slope, vegetation, fire characteristics, treatment combinations, weather patterns, and climate.

In the research we reviewed, sediment yield from pretreatment or control areas ranged from as low as 3 pounds per acre per year [3.36 kilograms (kg) per hectare per year] to as high as 12,500 pounds per acre per year [14 metric tonnes (Mg) per hectare per year] (table 12.3). Postburn sediment yield ranged from as low as 12 pounds per acre per year (13.5 kg per hectare per year) to as high as 98,160 pounds per acre per year (110 Mg per hectare per year). The lower values generally were associated with flatter land and lower severity fires. The higher values resulted from more severe fires on steeper slopes and from fires on areas with soils formed from decomposing granite, which erode readily.

When fire is used to convert brush to grass, it can have an unintended side effect: mass wasting, which can affect water quality. Work in California established the susceptibility of steep slopes to mass soil movement following conversion of brush to grass (Bailey and Rice 1969). These mass soil movements produce long-lasting changes. In one study, these same effects occurred on steep, forested slopes; especially after severe fires (Robichaud and Waldrop 1994) (table 12.3). These sediment yields are sufficient to generate concern about water turbidity, which was not measured directly.

Burned areas are sometimes seeded to rapidly establish plants or are given other treatments to quickly stabilize the soil. Following severe wildfire, the Forest Service and other land managers sometimes implement Burn Area Emergency Rehabilitation (BAER) treatments to reduce the risk of high runoff and sediment flows to vulnerable installations downstream such as drinking water intakes and reservoirs. In a review of literature and monitoring reports, Robichaud and others (in press) found that the effectiveness of the most widely used BAER practice, contour-felled log barriers, had not been systematically studied. The second most used BAER practice, postfire broadcast seeding with grasses, has been studied and the majority of studies found that this treatment did not significantly reduce erosion during the critical first 2 years after fire (Robichaud and others 2000). Effectiveness of contour felling has not been tested, and reseeding with grasses is not a reliable technique for erosion control after severe wildfire. Additionally, when an area is seeded with nonnative grass species, native plant species may be effectively excluded leading to questions about long-term stability (Tiedemann and Klock 1976).

Firelines, particularly those that are created by bulldozers, are important potential sources of suspended sediment and turbidity in streams for several reasons. First, some firelines are constructed in urgent circumstances, without adequate time to consider stream protection. Thus, they may provide direct channels for sediment into streams. Second, firelines may be difficult to stabilize with vegetation because much of the nutrient-rich surface soil is cast aside. Hence, they are likely to be slow to revegetate with perennial vegetation. Information on revegetating and stabilizing firelines is very limited. Two studies found application of seed and fertilizer is an effective way to protect firelines (Klock and others 1975, Tiedemann and Driver 1983). Klock and others (1975) demonstrated that seeding firelines with several species of introduced and native grasses produced up to 85 percent foliar cover within 2 years. In their area of nitrogen- and sulfur-limited soils, starter fertilizer containing nitrogen and sulfur substantially improved plant foliar cover and was considered to be essential for successful seeding.

Table 12.3—Sediment yield after fire alone or in combination with other treatments

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - - Pounds per acre per year - - - - - -		
Prescribed burn	Bluestem grasses	Upper Coastal Plain, northern Mississippi	74 – 84	1,542 – 5,759	Ursic 1969
Prescribed underburn	Post oak, hickories, blackjack oak	Upper Coastal Plain, northern Mississippi	148 – 300	868 – 1,179	Ursic 1970
Prescribed fire, pile, and burn	Juniper	Central Texas			Wright and others 1976
		3 to 4% slope	18	18	
		8 to 20% slope	146	504	
		37 to 61% slope	6	5,554	
Wildfire	Ponderosa pine	Northwestern Arizona	3	1,254	Campbell and others 1977
	Ponderosa pine, Douglas-fir	Eastern Washington	12 – 35	146 – 2,100	Helvey 1980, Helvey and others 1985
Prescribed underburn	Loblolly pine plantation	Upper Piedmont, South Carolina	19	20	Douglass and Van Lear 1983
Wildfire	Chaparral	Southern California	12,500	98,160	Wells 1986
Clearcut: broadcast burned, planted to Douglas-fir, wildfire	After clearcut: vegetation: tanoak, madrone, chinkapin, black oak, poison oak	Southern Oregon	80 [a]	55	Amaranthus 1989
Adjacent forest: wildfire	Douglas-fir overstory; tanoak, madrone, black oak understory		79	40	Amaranthus 1989
Clearcut and prescribed fire:	Oak spp., shortleaf pine	Northwestern South Carolina	Not known		Robichaud and Waldrop 1994
Low severity				12.1	
High severity				502	

[a] From October 13 to May 4, after September wildfire.

Temperature

Issues and Risks

Increases in streamwater temperature have important effects on aquatic habitat and stream and lake eutrophication. Eutrophication can adversely affect the color, taste, and smell of drinking water. See chapter 2 for temperature impacts on drinking water.

Findings from Studies

When riparian vegetation is removed by fire or other means, the stream surface is exposed to direct solar radiation, and stream temperatures increase (Levno and Rothacher 1969, Swift and Messer 1971). For example, clearcutting and slash burning increased stream temperatures by 13.0, 14.0, and 12.1 °F (7.2, 7.7, and 6.7 °C) in June, July, and August, with temperatures reaching a maximum of 75 °F (23.9 °C) in July (Levno and Rothacher 1969). Helvey (1972) found that during the first year after wildfire in eastern Washington, stream temperature increased 10 °F (5.6 °C). In southern Oregon, Amaranthus and others (1989) determined that temperatures increased 6, 11, and 18 °F (3.3, 6.1, and 10 °C), from a low temperature of 55 °F (12.8 °C) to a high temperature of 73 °F (22.8 °C) after a wildfire. These temperature changes have the potential to increase the rate of eutrophication if phosphate is present in abundance.

Chemical Water Quality

Several chemical constituents are likely to come from forest and rangeland burning. The primary ones of concern are nitrate (NO_3^{-1}) and nitrite (NO_2^{-1}). Sulfate, pH, total dissolved solids, chloride, iron, turbidity (discussed previously), and several other constituents can also be affected, as can color, taste, and smell (see chapter 2). Phosphate (P) can affect water quality because of its ability to affect color, taste, and smell by accelerating the eutrophication process.

To understand the influence of fire on water quality, it is important to understand some of the changes in plant, forest floor, and soil nutrients during and after the combustion process. Burning oxidizes organic material, resulting in direct loss of elements to the atmosphere as volatilized compounds above critical temperatures, as particulates are carried away in smoke, or elements are converted to oxides to the ash layer (DeBano and others 1998, Raison and others 1985, Tiedemann 1981) (fig. 12.2). Nitrogen, sulfur, and potassium are all susceptible to volatilization loss by burning (DeBano and others 1998, Raison and others 1985, Tiedemann 1987). Nitrogen is lost when temperatures reach 400 °F (204 °C) (DeBano and others 1998). At temperatures as low as 700 °F (371 °C), loss of sulfur can be substantial (Tiedemann 1987). As temperatures approach 1,475 °F (802 °C), virtually all nitrogen and sulfur are volatilized. At 1,430 °F (776 °C), phosphorus and potassium are volatilized. In ashes, relatively insoluble oxides of metallic cations, such as calcium, potassium, magnesium, and iron, react with water and carbon dioxide in the atmosphere and become more soluble (DeBano and others 1998, Tiedemann 1981) (fig. 12.2). This conversion increases potential for leaching loss of nutrients from the ash into and through the soil (DeBano and others 1998, Tiedemann 1981, Wells and others 1979). Nutrients in the ash are also susceptible to loss by surface erosion (Beschta 1990, DeBano and others 1998, Tiedemann 1981, Tiedemann and others 1979, Wells and others 1979).

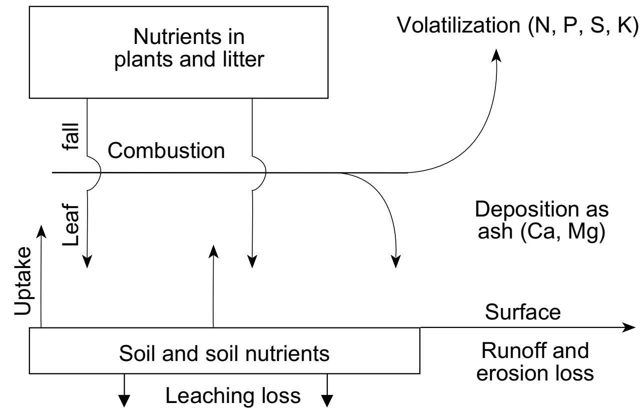

Figure 12.2—Possible pathways of plant- and litter-contained nutrients in response to combustion (Tiedemann 1981).

The potential for increased nitrate in streamflow occurs mainly because of accelerated mineralization and nitrification in soils after burning (Covington and Sackett 1986, 1992; DeBano and others 1998; Vitousek and Melillo 1979), as well as reduced plant demand (Vitousek and Melillo 1979). This effect is short-lived, usually lasting only a year or so (Monleon and others 1997).

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

Issues and Risks

The issue is whether forest or rangeland fires degrade the quality of source water for public consumption by the introduction of additional chemical constituents. The risk is when these additional chemical constituents—from a fire or from fertilizer applied to establish vegetation in the burn area—are combined with chemical constituents already present, the source water supply may be degraded.

Findings from Studies

Immediately after a fire, the pH of streams may be affected by direct ash deposition. In the first year after fire, increased pH of the soil (Wells and others 1979) may also contribute to increased streamwater pH. In all the studies we evaluated (table 12.4), only one reported a notable increase in pH values. During the first 8 months after the Entiat fires in eastern Washington, Tiedemann (1973) detected transient pH values up to 9.5. Two days after fertilization, they detected a transient pH value of 9.2. In most studies pH values were little changed by fire and fire-associated events.

Nitrogen

The forms of nitrogen that are of concern in drinking water after fire are nitrate and nitrite. Values for nitrate generally increased after fire but not to a level of concern (table 12.5), except in nitrogen-saturated areas (see chapter 3). Stream nitrate responses to prescribed fire are generally lower than for wildfire. In an undisturbed ponderosa pine and Gambel oak or both (*P. ponderosa* Dougl. ex Laws. and *Q. gambelii* Nutt. or both) watershed in Arizona, Gottfried and DeBano (1990) found that a fire resulted in only slight, but significant, increases in nitrate (table 12.5). Measures to protect streams and riparian areas during prescribed burns with unburned buffers could minimize effects of fire on stream chemistry.

The most striking response of nitrate concentration in streamflow after wildfire (table 12.5) was observed in southern California (Riggan and others 1994). Moderate burning resulted in a maximum nitrate concentration of 9.5 ppm, while severe burning resulted in a maximum concentration of 15.3 ppm in streamflow, compared to 2.5 ppm in streamflow from an unburned control watershed. The concentration of 15.3 ppm is above maximum contaminant level for drinking water of 10 ppm (chapter 2, table 2.3). Chronic atmospheric deposition of nitrogen pollutants on these watersheds, which are east of Los Angeles, CA, have caused their soils to become nitrogen saturated. Beschta (1990) reached the same conclusion in his assessment of streamflow nitrate responses to fire and associated treatments. Fenn and others (1998) have discussed excess nitrogen in ecosystems in North America. These excess levels can lead to leaching of nitrate, which ultimately can find its way into streamwater (see chapter 3).

Fertilization after fire resulted in higher concentrations of nitrate than fire alone (table 12.5) (Tiedemann 1973, Tiedemann and others 1978). Nonetheless, Tiedemann (1973) concluded that neither fire nor nitrogen fertilization at levels less than 54 pounds per acre (60.5 kg per hectare) of elemental nitrogen would probably have adverse effects on nitrate concentrations in drinking water. Their research was done in an area with nitrogen-limiting soils. In areas experiencing nitrogen saturation, nitrogen fertilization may aggravate nitrate levels in water and is not likely to stimulate revegetation.

Nitrite was reported by itself, rather than in combination with nitrate, in only two studies that we found. At concentrations >1 ppm, nitrite can lead to serious illness in infants (chapter 2, table 2.3). At the Lexington Reservoir, Santa Clara County, CA, Taylor and others (1993) found nitrite levels of 0.03 ppm after the watershed above the reservoir was burned, while control levels were 0.01 ppm. Tiedemann (1973) reported that nitrite concentrations were below the levels of detection. The concentrations found do not appear to be a concern.

Fire retardants containing nitrogen have the potential to affect the quality of drinking water, but research on the application of retardants to streams has focused on the effects on fish and aquatic habitat (Buhl and Hamilton 1998; Gaikowski and others 1996; McDonald and others 1996, 1997; Norris and Webb 1989; Norris and others 1978). Several in vitro research projects evaluated the toxicity to stream organisms of some retardant formulations. The tested compounds were nonfoam retardants containing sulfate, phosphate, and ammonium compounds; a retardant containing ammonium and phosphate compounds; and two foam suppressant compounds (Buhl and Hamilton 1998; Gaikowski and others 1996; McDonald and others 1996, 1997). Concentrations of nitrate rose from 0.08 to 3.93 ppm after adding the nonfoam retardants. In addition, they found

Table 12.4—The pH in water after fire alone or in combination with other treatments usually remains fairly constant

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - - - - - *pH*	- - - - - - - - -	
Wildfire	Ponderosa pine, Douglas-fir	Eastern Washington	None given	7.2 – 8.5	Tiedemann 1973
Wildfire and N fertilization			None given	7.1 – 9.5[a]	
Wildfire and N fertilization	Mixed conifer, shrub	Central Sierra Nevada Mountains, California	~7.0 – 6.2[b]	~7.0 – 6.6[b]	Hoffman and Ferreira 1976
Pile, burn	Juniper	Central Texas			
		3 to 4% slope	7.3	7.3	Wright and others 1976
		8 to 20% slope	7.6	7.7	
		37 to 61% slope	7.4	7.7	
Wildfire	Pine, spruce, fir, aspen, birch[c]	Northeastern Minnesota lakes	6.2	6.1 – 6.3	Tarapchak and Wright 1977
Wildfire Wildfire and N fertilization	Ponderosa pine, Douglas-fir	Eastern Washington	7.4 – 7.6	7.4 – 7.6	Tiedemann and others 1978
Prescribed fire	Ponderosa pine	Central Arizona	6.2	6.4	Sims and others 1981
Pile, burn, and seed	Juniper	Central Texas	7.1	7.3	Wright and others 1982
Clearcut, slash broadcast burned	Western hemlock, western red cedar, Douglas-fir	Western British Columbia	6.8	7.8	Feller and Kimmins 1984
Yellowstone wildfires	Subalpine lake	Yellowstone Lake, Yellowstone National Park, WY	7.4	7.5	Lathrop 1994

~ = About or approximately.
[a] Transient pH value of 9.5 was observed second day after urea fertilization.
[b] From May to July during the summer following the August fire.
[c] Cited in Wright and Watts 1969.

Table 12.5—Maximum nitrate-nitrogen concentration in water after fire alone or in combination with other treatments

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - *Parts per million* - - - - -		
Clearcut, slash burned	Douglas-fir	Western Oregon	0.1	0.43	Fredriksen 1971
Wildfire	Ponderosa pine	Eastern Washington	.016[a]	.042	Tiedemann 1973
Wildfire and nitrogen fertilization	Douglas-fir		.005	.310[a]	
Wildfire	Mixed conifer, shrub	Central Sierra Nevada Mountains, California	~.6[a b]	~.12	Hoffman and Ferreira 1976
	Ponderosa pine	Northwestern Arizona	.086	.212	Campbell and others 1977
	Pine, spruce, fir, aspen, birch[c]	Northeastern Minnesota lakes	.17	.08 – .17	Tarapchak and Wright 1977
Wildfire	Ponderosa pine	Eastern Washington	< .016[a]	.56	Tiedemann and others 1978
Wildfire and nitrogen fertilization	Douglas-fir		< .016[a]	.54 –1.47	
Prescribed fire	Pine forest or not given	Lower Coastal Plain, South Carolina	[d]	.02	Richter and others 1982
Prescribed underburn	Loblolly pine plantation	Upper Piedmont, South Carolina	.05	.05	Douglass and Van Lear 1983
Clearcut, slash broadcast burned	Douglas-fir, ponderosa pine	Southern Idaho	.02	.05	Clayton and Kennedy 1985
Prescribed burn, moderate	Ponderosa pine, gambel oak	Central Arizona	0.0013[a]	0.0029	Gottfried and DeBano 1990
Wildfire	Chaparral	Lexington Reservoir, Santa Clara County, CA	.02	.04	Taylor and others 1993
Prescribed underburn	White fir, giant sequoia, red fir, sugar pine, Jeffrey pine	Sequoia National Park, CA	.001 – .005	.010 – .394	Chorover and others 1994
Prescribed broadcast:	Chaparral	Southern California			Riggan and others 1994
Moderate burn			2.5	9.5	
Severe burn			2.5	15.3	
Wildfire	Lodgepole pine, Douglas-fir, ponderosa pine, western larch	Glacier National Park, MT	< .040	.124 – .312	Hauer and Spencer 1998

~ = About or approximately.
[a] Maximum level attained.
[b] Mean concentration from May to July after August fire.
[c] Cited in Wright and Watts (1969).
[d] Pretreatment not significantly different from posttreatment.

in vitro nitrite reached concentrations as high as 33.2 ppm. Accidental deposition of retardants in streams has produced values of nitrate and ammonia sufficiently high to be of concern in drinking water.[2] Great caution needs to be exerted to keep retardant chemicals out of streams that are public drinking water sources.

Phosphorus

Phosphate, as a component of fire retardants, can lead to eutrophication. See chapter 3 for discussion of phosphorus impacts on drinking water. Prior to wildfire, phosphate concentrations ranged from 0.007 ppm to 0.17 ppm (Hoffman and Ferreira 1976, Tiedemann and others 1978, Wright and others 1976). After wildfire, prescribed fire, or clearcutting followed by broadcast burning, phosphate concentrations stayed the same or increased only as high as 0.2 ppm (Longstreth and Patten 1975). Any phosphorus added to the stream system may have been taken up by the aquatic organisms and, therefore, little increase was detectable. We found no reports of changes in phosphate concentration as the result of an inadvertent application of retardant directly into a stream.

Sulfur

The sulfate ion is relatively mobile in soil water systems (Johnson and Cole 1977). Although not as well studied as those for nitrogen, the mineralization processes for sulfur are similar. In streamwater from wildland watersheds, observed levels of sulfate are usually low (table 12.6). Control or prefire values range from as low as 1.17 ppm to as high as 66 ppm, while postfire values range from 1.7 ppm to a high of 80.7 ppm, well below the recommended secondary drinking water standard (250 ppm) (table 2.4).

Chloride

Chloride response to fire and clearcutting plus fire has been documented in several studies, and all responses are low (table 12.7). Chloride concentrations in control or prefire samples ranged from 0.49 to 6.4 ppm, and the chloride concentration in postfire samples ranged from 0.40 to 7.1 ppm (Lathrop 1994), well below the recommended secondary drinking water standard (250 ppm) (table 2.4). Lewis Lake in Yellowstone National Park, WY, with its large volume of water, had the highest chloride values for both the prefire and postfire periods among the data examined.

Total Dissolved Solids

Only two studies reported total dissolved solids; many other studies measured some of the constituents of total dissolved solids but not total dissolved solids per se. Hoffman and Ferreira (1976) detected a total dissolved solids concentration of about 11 ppm in the control area and 13 ppm in the burned area, which had been a mixed conifer and shrub stand in Kings Canyon National Park, CA. Lathrop (1994) found Yellowstone Lake in Yellowstone National Park and Lewis Lake had pretreatment total dissolved solids concentrations of 65.8 and 70 ppm. The total dissolved solids concentrations after the fires were 64.8 and 76 ppm, well below the recommended secondary drinking water standard (500 ppm) (table 2.4).

Trace Elements

Fredriksen's (1971) results raise a question about how well we understand the responses of micronutrients or trace elements to fire or to fire after clearcutting. In his stream chemistry profile after clearcutting and broadcast burning, he documented a maximum concentration of manganese of 0.44 ppm, exceeding the recommended secondary drinking water standard (0.05 ppm) (table 2.4), which may raise palatability issues but is not a health risk. There are established drinking water standards for 14 additional trace constituents, including heavy metals. Information on the effects of these elements after a forest or rangeland fire on drinking water quality is lacking.

Effects on Ground Water

Little research has been conducted on the effects of fire, fire suppression, and fire rehabilitation activities on ground water quality. It is reasonable to expect that fire will have little effect on ground water quality. A possible, but unlikely, scenario would be a fire followed by an intense long-duration precipitation event sufficient to cause major flooding, which could contaminate ground water. In such a case, the fire sets the stage for contamination of the ground water source.

[2] Labat-Anderson Incorporated. 1994. Chemicals used in wildland fire suppression: a risk assessment. Prepared for: Fire and Aviation Management, U.S. Department of Agriculture, Forest Service. Contract 53-3187-9-30; Task 93-02. 187 p. Prepared by: Labat-Anderson Incorporated, 2200 Clarendon Boulevard, Suite 900, Arlington, VA 22202.

Table 12.6—Sulfate concentration in water after fire alone

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - *Parts per million* - - - -		
Wildfire	Taiga	Interior Alaska	7.12 – 66	8.3 –80.7	Lotspeich and others 1970
	Mixed conifer, shrub	Central Sierra Nevada Mountains, CA	1.5	1.7	Hoffman and Ferreira 1976
	Pine, spruce, fir, aspen, birch[a]	Northeastern Minnesota lakes	1.17	1.79 – 1.86	Tarapchak and Wright 1977
Prescribed underburn	White fir, giant sequoia, red fir, sugar pine, Jeffrey pine	Sequoia National Park, CA, Log Creek, control	.26	.37, .30, .45[b]	Chorover and others 1994
	White fir, fewer Giant sequoia	Tharp's Creek, burn	.24	9.68, 1.32, 2.15[b]	
Yellowstone wildfires	Subalpine lakes	Yellowstone Lake	8.9	6.4[c]	Lathrop 1994
		Lewis Lake, Yellowstone National Park, WY	4.0	3.0	

[a] Cited in Wright and Watts 1969.
[b] Postburn years one, two, and three, in sequence.
[c] Average of reported median values from four areas of Yellowstone Lake.

Table 12.7—Chloride concentration in water after fire alone

Treatment	Habitat	Location	Pretreatment or control	Posttreatment	Reference
			- - - - - *Parts per million* - - - -		
Wildfire	Taiga	Interior Alaska	0.9 – 5.0	1.2 – 4.6	Lotspeich and others 1970
	Mixed conifer, shrub	Central Sierra Nevada Mountains, CA	.6	1.0	Hoffman and Ferreira 1976
	Pine, spruce, fir, aspen, birch[a]	Northeastern Minnesota lakes	.80 – .89	1.24	Tarapchak and Wright 1977
Prescribed underburn	White fir, giant sequoia, red fir, sugar pine, Jeffrey pine	Sequoia National Park, CA	.49 – .56	.40 – 2.78	Chorover and others 1994
Yellowstone wildfires	Subalpine lakes	Yellowstone Lake	5.1	3.6	Lathrop 1994
		Lewis Lake, Yellowstone National Park, WY	6.4	7.1	

[a] Cited in Wright and Watts 1969.

Reliability and Limitations of Findings

The results of research on the effects of fire on drinking water quality are strong and consistent, especially from the Pacific Northwest, the Rocky Mountains, and the Southwest. The results indicate that the effects of fire and fire management practices on water quality are similar within each of these three large areas. Data from the Southeast are somewhat more limited in spite of the region's extensive prescribed fire program. In Alaska, water-quality research after fire has been minimal, even though the area has many wildfires.

With the changes in pH, nitrate, sulfate, and chloride so consistently small, a land manager can safely assume that effects will be similar to those found in the literature, if the treatments and fire severity, slope, and soil and vegetation types are comparable. In areas likely to be nitrogen saturated, such as areas of high soil concentrations of nitrogen from chronic atmospheric deposition, nitrate concentrations in streamwater after a fire may exceed the established maximum contaminant level of 10 ppm (chapter 2, table 2.3). In areas of suspected nitrogen saturation, common sense tells us that nitrogen-containing fertilizer should not be applied. Application of nitrogen fertilizer would exacerbate the risk of degrading source water supplies. See chapter 3 for more discussion of nitrogen saturation.

Results of previous wildland fires can be used as a basis for estimating the effects of new fires on drinking water quality. Fires need to be of the same type; that is, previous wildfires should be used as the comparison basis for new wildfires, and previous prescribed fires as the basis for new prescribed fires. The more factors, such as slope and vegetation, among others, that match between the previously documented fires and the new fires, the closer the approximation. Nevertheless, results and predictions based on limited data must be used cautiously. In two studies (Beschta 1980, Helvey and others 1985), the researchers specifically caution against extrapolating results of turbidity and sediment research beyond the watersheds in which the research was conducted.

Research Needs

1. Research methods need to be carefully selected for measurements of sediment. Suspended sediment concentration is used in some studies while turbidity, the standard measurement, is used in others. At this time, regression relationships between sediment, in parts per million, and turbidity, in NTU's, need to be developed for each individual watershed or stream system. Interpretation of future research results will be facilitated when all measurements are reported in the standard NTU's.
2. Areas with chronic atmospheric deposition, such as those studied by Riggan and others (1994), need further research into the relations between fire and nitrogen release into streams.
3. We have little information on the abundance of trace elements (micronutrients) after fire. When elements, such as lead, copper, fluoride, manganese, iron, zinc, and mercury, among others, are above certain levels, they are important potential contaminants in drinking water supplies. We do not understand the effects of fire in combination with other treatments on micronutrients. Effects may be particularly important for some of the heavy-metal trace elements.
4. The inadvertent application of fire retardants directly into a stream can produce increased levels of nitrate and possibly sulfate, phosphate, and some trace elements. Information is needed about the potential effects of specific retardants on drinking water quality.
5. The BAER practices, particularly the use of contour-felled erosion barriers, need to be systematically studied to determine their effectiveness for reducing storm runoff, erosion, and sediment movement, which pose a risk to the quality of source water for public water supplies.

Key Points

1. When a wildland fire occurs, the principal concerns for change in drinking water quality are: (1) the introduction of sediment; and (2) the potential introduction of nitrates, especially if in areas with chronic atmospheric deposition.
2. As we considered the above types of fire effects on drinking water, several concepts important to the land manager became apparent. The magnitude of the effects of fire on water quality is primarily driven by fire severity, and not necessarily by fire intensity. Fire severity is a qualitative term describing the amount of fuel consumed, while fire intensity is a quantitative measure of the rate of heat release. In other words, the more severe the fire the greater the amount of fuel consumed and nutrients released and the more susceptible the site is to erosion of soil and nutrients into the stream where it could potentially affect water quality. Wildfires usually are more severe than prescribed fires, and, as a result, they are more likely to produce significant effects on water quality. On the other hand, prescribed fires are designed to be less severe and would be expected to produce less effect on water quality. Use of prescribed fire allows the manager the opportunity to control the

severity of the fire and to avoid creating large areas burned at high severity. The degree of fire severity is also related to the vegetation type. For example, in grasslands the differences between prescribed fire and wildfire are probably small. In forested environments, the magnitude of the effects of fire on water quality will probably be much lower after a prescribed fire than after a wildfire because a larger amount of fuel may be consumed in a wildfire. Canopy-consuming wildfires would be expected to be of the most concern to managers because of the loss of canopy coupled with the destruction of soil aggregates. These losses present the worst-case scenario in terms of water quality. The differences between wild and prescribed fire in shrublands are probably intermediate between those seen in grass and forest environments.

3. Another important determinant of the magnitude of the effects of fire on water quality is slope. Steepness of the slope has a significant influence on movement of soil and nutrients into stream channels where it can affect water quality. Wright and others (1976) found that as slope increased in a prescribed fire, erosion from slopes is accelerated. If at all possible, the vegetative canopy on steep, erodible slopes needs to be maintained, particularly if adequate streamside buffer strips do not exist to trap the large amounts of sediment and nutrients than can be transported quickly into the stream channel. It is important to maintain streamside buffer strips whenever possible, especially when developing prescribed fire plans. These buffer strips will capture much of the sediment and nutrients from burned upslope areas.

4. Two more concerns, which are more site-specific, deal with soils. Both the general type of soil and a soil's propensity to develop water repellency can be determinants of the magnitude of the effects of fire on drinking water quality. When sandy soils are burned, nutrient transport and loss are rapid. These soils do not have the ability to capture and hold nutrients, but, rather, allow the nutrients to move into the ground water and eventually into nearby streams. Additionally, in areas with sandy soils, which contain few nutrients, most of the nutrient capital is stored aboveground. A severe fire volatilizes many of these nutrients, impoverishing the site, while adding to the nutrient load in streams. Prescribed fires in these areas need to be very carefully planned to retain as many nutrients on site as possible through the use of low-severity fires.

5. If a site is close to nitrogen saturation, it is possible to exceed maximum contamination levels for drinking water of nitrate (10 ppm) after a severe fire. Such areas should not have nitrogen-containing fertilizer applied after the fire. See chapter 3 for more discussion of nitrogen saturation.

6. The propensity for a site to develop water repellency after fire must be considered. Water-repellent soils do not allow precipitation to penetrate down into the soil and, therefore, are conducive to erosion. Such sites can put large amounts of sediment and nutrients into surface water.

7. Finally, heavy rain on recently burned land can seriously degrade water quality. The effects of the Buffalo Creek wildfire on the water supply for Denver, noted in the first paragraph of this chapter, demonstrated these effects on a water supply. Severe erosion and runoff are not limited to wildfire sites alone. If the storm delivers large amounts of precipitation or is sufficiently intense, accelerated erosion and runoff can occur after a carefully planned prescribed fire. Conversely, if below-average precipitation occurs after a wildfire, there may not be a substantial increase in erosion and runoff.

8. The land manager can influence the effects of fire on drinking water quality by careful prescribed burning. Limiting fire severity, avoiding burning on steep slopes, and limiting burning on sandy or potentially water-repellent soils will reduce the magnitude of the effects of fire on water quality.

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Chapter 13

Pesticides

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Introduction

On forest and grassland, management often must protect desirable vegetation from pathogens, competing vegetation, insects, and animals. Vegetation also is managed to clear road and utility rights-of-way, to improve recreation areas and wildlife habitat, and to control noxious weeds. Pesticides offer inexpensive and effective ways of getting these jobs done. The Forest Service requires: (1) training of personnel who recommend and use pesticides, (2) applicator certification, and (3) safety plans to assure the safety of personnel and the protection of environmental values like drinking water quality. Nonnational forest land is treated with pesticides for many of the same purposes, but often more intensively.

The Federal Insecticide, Fungicide, and Rodenticide Act, as amended (FIFRA) (Public Law 92–516, and 40 CFR 158) allows the registration of pesticides for use in the United States. The registration process is an extraordinary one that requires years of testing before sufficient efficacy, environmental safety, toxicology, and public safety data can be collected and evaluated in the support of registration of a new pesticide. While this process is designed to assure safety, new and old pesticides, following registration, continue to be studied by researchers in private, State, and Federal agencies in an effort to identify any potential environmental or toxicological problems. An integral part of protecting public health and environmental values during pesticide use is the requirement that they must be applied according to directions approved by the U.S. Environmental Protection Agency (EPA) and included on the label of every registered pesticide. Under FIFRA, pesticide labels are legally binding documents, and any infraction of the directions for application is a violation of law. Users of pesticides must exercise extreme caution in following label directions and must also exercise good judgement, especially when pesticide use is planned in an area near municipal water supplies. In addition, pesticide users must provide adequate handling facilities for mixing and storage and be well prepared to deal with spills.

To meet the minimum requirements of FIFRA at the State level, the EPA has established and maintains cooperative enforcement agreements for pesticide use inspections, producer establishment inspections, marketplace surveillance, applicator certification, and experimental use inspections. State government is responsible for (1) certification of pesticide applicators, (2) enforcement of FIFRA pesticide use regulations and inspections, (3) endangered species considerations, (4) worker protection, and (5) ground water protection.

When forestry pesticides are used near water on Federal, State, or privately owned land, buffer zones are left between the treated areas and the water resource (see chapter 5). The width of the buffer varies with site conditions, site sensitivity, and local or State recommendations. National forests in some States use more conservative buffers than those recommended by the State. Comerford and others (1992) have reviewed many agricultural studies in an attempt to draw inferences regarding effectiveness of buffer strips in mediating stream contamination. However, relatively little research data are available on effectiveness of buffers on forest sites. It, therefore, is not possible to determine the minimum buffer width to protect streams from either pesticide or sediment contamination.

Issues and Risks, Pesticide Application

Approximately 16 percent of the 3.6 million square miles of land in the United States is treated with pesticides annually (Pimentel and Levitan 1986). The most intensive use of pesticides occurs on land occupied by households. Household tracts account for only 0.4 percent of all land but receive 12 percent of all pesticides used in the United States. Agricultural land (52 percent of all land) is the next most intensively treated, receiving 75 percent of all pesticides used. Government and industrial land (16 percent of all land) receives 12 percent of all pesticides. The least intensive use of pesticides occurs on forest land (32 percent of the land). Pimentel and Levitan (1986) point out that forest land receives only 1 percent of all pesticides used in the United States and that <1 percent of all forest land is treated annually.

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A wide variety of pesticides are used on forest and grasslands. Table 13.1 lists these pesticides and the purposes for which they are used. Maximum Contaminant Levels (MCL's), established by EPA, are listed in chapter 2, table 2.3.

Biological control agents, including *Bacillus thuringiensis* (Bt) and nucleopolyhedrosis virus (Npv), are used for control of western spruce budworms and gypsy moths. While these are pathogens of insects, they have no known impacts on drinking water quality.

Plant pathogens represent a potential problem throughout forest and rangeland ecosystems, but their destructive impacts are most severe in seedling nurseries and seed storage facilities. Fungicides and fumigants are used to control these pathogens on seeds, in seedling nurseries, in greenhouses, in seed tree nurseries, and on individual trees. In 1997, the Forest Service treated 35 greenhouses with fungicides and fumigants including benomyl, chlorothalonil, dicloran, iprodione, metalaxyl, propiconazole, thiophanate methyl, and triadimefon. Most fungicide and fumigant use occurs on small acreages in nurseries for disease control.

Small amounts of strychnine and putrescent egg solids were used over extensive acreages for animal damage control. While very small amounts of strychnine were used over vast acreages, it is very toxic (table 13.2). Putrescent egg solids, by comparison, are derived from food products and the EPA has waived toxicology requirements. These two products accounted for more than 96 percent of the active ingredients used in protection of vegetation from animals. Insect control relied mainly on biological agents, but some insecticides and oils were used. The insecticides included carbaryl and chlorpyrifos. Dormant oil was used for control of a variety of insects and their eggs. These three (carbaryl, chlorpyrifos, and dormant oil) represent 94.6 percent of all chemical insecticides used on National Forest System (NFS) land.

Vegetation management is frequently taken to mean the control of competing vegetation in timber management programs. On NFS land, more than three times as much land was treated for protection of vegetation from animals and insects and to control noxious weeds, than for control of competing vegetation in timber management programs in fiscal year 1997 (U.S. Department of Agriculture, Forest Service 1998). Competing vegetation can be controlled with herbicides, algicides, and plant growth regulators. Table 13.1 shows management objectives for herbicide use on NFS land in fiscal year 1997. Many acres are treated for timber management, principally planting site preparation and release of crop trees. Such treatments usually occur only once or twice over a rotation. Rotation length depends on tree species, site productivity, and management objectives. The rotation may be as short as 20, or longer than 150 years. Thus, herbicides are used in timber management only once or twice in 20 or more years. Treatments were for site preparation, conifer release, and hardwood release.

Noxious weed control is often accomplished by treatment with herbicides. Noxious weeds are usually nonnative plants that, lacking natural controls, spread quickly and take over or ruin habitat for native plants. They generally possess one or more of the following characteristics: aggressive and difficult to manage, poisonous, toxic, parasitic, and a carrier or host of serious insects or disease. There are 74 terrestrial species on the Federal noxious weed list, including kudzu. The frequency of noxious-weed treatment varies by species. In southern forests, kudzu requires annual treatment over several years for effective control. Typically, attempts to control noxious weeds do not eradicate them, but bring them under enough control to reduce immediate problems. Timber management (45.2 percent) and noxious weed control (48.8 percent) accounted for 94 percent of all acres treated with herbicides.

Protecting forests and seedlings from animal pests is the single largest component of the vegetation management program. Rabbits and deer were the most common target mammals, while western spruce budworms and gypsy moths were the principal target insects.

One major issue with pesticide use is the impact on drinking water quality. To adversely impact drinking water, pesticides must (1) be harmful to humans, and (2) reach drinking water at concentrations exceeding toxic levels for humans.

Issues and Risks, Toxicity

The toxicity of a chemical is a measure of its ability to harm individuals of the species under consideration. This harm may come from interference with biochemical processes, interruption of enzyme function, or organ damage. Toxicity may be expressed in many ways. Probably the best known term is LD_{50}, the dose at which 50 percent of the test animals are killed. More useful terms have come into popular usage in the last decade: no observed effect level (NOEL), no observed adverse effect level (NOAEL), lowest observed adverse effect level (LOAEL), reference dose (RfD), and, relating specifically to water, the health advisory level (HA or HAL). The EPA uses these terms extensively in risk assessment programs to indicate levels of exposure deemed safe for humans, including sensitive individuals. They are derived from toxicological test data and have built-in safety factors ranging upward from 10, depending on EPA's evaluation of the reliability of the test data.

Table 13.1—Management uses of pesticides commonly used on national forests

Pesticide	Vegetation management use(s)
2, 4-D	Housekeeping and facilities maintenance, noxious weed control, nursery weed control, recreation improvement, right-of-way vegetation management, seed orchard protectio agricultural weed control, other vegetation management
Bacillus thuringiensis	Insect suppression
Borax	Disease control
Carbaryl	Insect suppression in the field and in greenhouses, nursery insect control
Chloropicrin	Nursery disease control
Chlorpyrifos	Housekeeping and facilities maintenance, insect control, nursery insect control
Clopyralid	Housekeeping and facilities maintenance, noxious weed control, nursery weed control, right-of-way vegetation management, wildlife habitat improvement
Dazomet	Fungus control, nursery disease control, soil fumigation
Dicamba	Noxious weed control, other vegetation management
Dormant oil	Insect control
Hexazinone	Wildlife habitat improvement, site preparation, conifer release
Imazapyr	Conifer release, hardwood release, hardwood control, noxious weed control, site preparation
Methyl bromide	Nursery disease control, soil fumigation
Metsulfuron	Noxious weed control
Nucleopolyhedrosis virus	Insect suppression
Picloram	Noxious weed control, right-of-way vegetation management, weed control, wildlife habitat improvement
Putrescent egg solids	Animal damage control
Strychnine	Animal damage control, seed orchard protection
Thiram	Animal damage control, fungus control, nursery disease control
Triclopyr	Conifer release, hardwood control, hardwood release, noxious weed control, recreation improvement, right-of-way vegetation management, seed orchard protection, site preparation, thinning, general weed control, wildlife habitat improvement
Zinc phosphide	Animal damage control

The NOEL is determined from animal studies in which a range of doses is given daily; some doses cause adverse effects and others do not (U.S. EPA 1993). The NOAEL is derived from the test data where all doses have some effect, but some of the observed effects are not considered adverse to health. When EPA has data from a number of these tests, the lowest NOEL or NOAEL is divided by a safety factor of at least 100 to determine the RfD. The RfD is an estimate of a daily exposure to humans that is likely to be without an appreciable risk of deleterious effects during a lifetime.

Drinking water standards are calculated for humans by assuming that an adult weighs 155 pounds and consumes 2 pints of water per day, and a child weighs 22 pounds and consumes 1 pint of water per day over the period of exposure. The HAL's are calculated for 1 day, 10 days, longer term (10 percent of life expectancy), or lifetimes (70 years) by dividing the NOAEL or LOAEL by a safety factor and multiplying the resulting value by the ratio of body weight to amount of water consumed daily (U.S. EPA 1993). The safety factor can range from as low as 1, but is rarely <10, and goes as high as 10,000, depending on the available

Table 13.2—Estimates of safe levels for daily exposure to the 20 pesticides most used on National Forest System lands in fiscal year 1997 in the vegetation management program

Pesticide	RfD	NOEL	NOAEL	Lifetime HAL	Reference
	- - - - - *Milligrams per kilogram* - - - - -			*mg/L*	
Borax	0.09	NA	8.8	0.60[a]	U.S. EPA 1990
Carbaryl	.1	NA	9.6	.700	U.S. EPA 1989
Chloropicrin[b]	NA	NA	NA	NA	
Clopyralid	NA	NA	NA	NA	
Chlorpyrifos	.003	0.03	NA	.020	U.S. EPA 1993
2,4-D[c]	.01	NA	1	.070	U.S. EPA 1989
Dazomet[b]	NA	NA	NA	NA	
Dicamba	.03	NA	3	.200	U.S. EPA 1989
Dormant oil	NA	NA	NA	NA	
Glyphosate[c]	.1	20	NA	.700	U.S. EPA 1989
Hexazinone	.05	5	NA	.400	U.S. EPA 1996
Imazapyr	NA	250	NA	NA	U.S. EPA 1997
Methyl bromide	.0014	NA	1.4	.010	U.S. EPA 1990
Metsulfuron	.25	25	NA	NA	U.S. EPA 1988b
Picloram[c]	.007	7	NA	.500	U.S. EPA 1988a
Putrescent egg solids	NA[d]	NA	NA	NA	
Strychnine	.0003	None	None	NA	U.S. EPA 1998a
Thiram	.005	5	NA	NA	U.S. EPA 1992
Triclopyr	.05	5	NA	NA	U.S. EPA 1998b
Zinc phosphide	.0003	None	None	NA	U.S. EPA 1998a

HAL = health advisory level; NA = not available; NOAEL = no observed adverse effect level; NOEL = no observed effect level; RfD = reference dose.
[a] HAL for elemental boron.
[b] These fumigants are not expected to get into water.
[c] Maximum contaminant levels for glyphosate (0.700 mg per liter), 2,4-D (0.070 mg per liter), and picloram (0.500 mg per liter) are discussed in chapter 2.
[d] Made from food products; toxicology was waived by U.S. Environmental Protection Agency.

toxicological data. A safety factor of 10 is used when good NOAEL data are based on human exposures and are supported by chronic or subchronic data in other species. When NOAEL's are available for one or more animal species but not humans and good data for LOAEL in humans is available, a safety factor of 100 is used. When good chronic data are available identifying an LOAEL but not an NOAEL for one or more animal species, a safety factor of 1,000 is used. For situations where good chronic data are absent, but subchronic data identify an LOAEL but not an NOAEL, the safety factor of 10,000 is used. The EPA's estimates of safe levels for daily exposure to the pesticides most widely used on NFS land are summarized in table 13.2. Of the pesticides listed in table 13.2, elemental boron (potentially from borax) and methyl bromide are listed in EPA's drinking water contaminant candidate list for consideration for possible regulation. The MCL's have been established for 2,4-D [0.070 milligrams (mg) per liter], glyphosate (0.700 mg per liter), and picloram (0.500 mg per liter) and these are the same as the already established lifetime HAL's (table 13.2). Information on specific pesticides can be retrieved from the National Pesticides Telecommunication Network at http://ace.orst.edu/info/nptn, EPA site at http://www.epa.gov/epahome/search.html, Extension Toxicology Network at http://ace.orst.edu/info/extoxnet, Material Data Safety Sheets at http://siri.uvm.edu/msds, USDA Forest Service at http://www.fs.fed.us/foresthealth/pesticide, and many others.

Findings from Studies

Pesticides used by the NFS in vegetation management are used around the World in agricultural, forest, range, and urban applications. Some have been found in surface water, shallow ground water, and even in shallow wells (<30 ft), but in concentrations far below levels harmful to human health, and the occurrence is infrequent. Table 13.3 summarizes reports of pesticides from table 13.2 that have been detected in water in the United States.

Larson and others (1997) summarized the results of 236 studies throughout the United States on pesticide contamination of surface water by listing the maximum observed concentrations from each study. These studies were located principally around large river drainage basins and, therefore, represent cumulative pesticide contributions from a wide variety of uses. Monitoring results were reported for 52 pesticides approved for agricultural, urban, and forestry use and their metabolic byproducts. Of the pesticides listed in table 13.2, only six were reported to be present in surface water by Larson and others (1997). They were carbaryl, 1 report; hexazinone, 1 report; chlorpyrifos, 3 reports; picloram, 4 reports; dicamba, 5 reports; and 2,4-D, 24 reports. None of the reported concentrations exceeded EPA safe levels for human health except where application included placement directly in stream channels and most were <0.002 mg per liter. It is important to recognize that surface water is not necessarily drinking water. The studies summarized by Larson and others (1997) dealt with surface water, principally in lakes, reservoirs, and rivers, which would be treated prior to use for drinking. Thus, use of these pesticides according to label directions has not resulted in impairment of drinking water.

Reports of pesticide contamination of water are usually from agricultural (Kolpin and others 1997, Koterba and others 1993) or urban applications (Bruce and McMahon 1996), but the potential exists for contamination from forest vegetation management. Water from forests is generally much less contaminated than water from other land uses. Several studies on forest sites listed in table 13.3 present data for water collected directly from treated areas. The concentration of pesticides can appear quite high compared to samples taken from large rivers and lakes. Pesticide concentrations are greatly reduced by dilution as they move from the treated sites to downstream locations. Degradation of pesticides by biological, hydrolytic, and photolytic routes also contributes to downstream reductions in pesticide concentrations.

From 1985–87, Cavalier and others (1989) monitored 119 wells, springs, and municipal water supplies for occurrence of pesticides in drinking water throughout the State of Arkansas. Monitored wells were generally located in the eastern portion of Arkansas, but eight wells were located in the Ouachita National Forest. Only sites considered highly susceptible to contamination from pesticide use were monitored, and these included domestic, municipal, and irrigation wells. Detection limits for the three forestry pesticides monitored (2,4-D, hexazinone, and picloram) ranged from 70 to 800 times lower than their HAL's. They did not detect well water contamination from any of the 18 pesticides for which they monitored. Failure to detect pesticides in these wells believed to be at high risk for contamination is a very strong indicator that ground water is not at risk from forestry pesticides used according to label directions.

Michael and Neary (1993) reported on 23 studies conducted on industrial forests in the South in which whole watersheds received herbicide treatment. Water flowing from the sites was sampled near the downstream edge of the treatments. The watersheds were relatively small (<300 acres) and the ephemeral to first-order streams draining these watersheds were too small to be public drinking water sources, but their flow reached downstream reservoirs. The maximum observed hexazinone, imazapyr, picloram, and sulfometuron concentrations in streams on these treated sites did not exceed HAL's, except for one case in which hexazinone was experimentally applied directly to the stream channel. Even in this case in which hexazinone was applied directly to the stream at a very high rate, drinking water standards were exceeded for only a few hours. In another study, picloram was accidentally applied directly to streams, but maximum picloram concentrations did not exceed HAL's during the year after application.

Bush and others (1990) reported on use of hexazinone on two Coastal Plain sites (deep sand and sandy loam soils) that were monitored for impacts on ground water. Hexazinone was not detected in ground water at the South Carolina site for 2 years after application. In Florida, hexazinone was found infrequently in shallow test wells at concentrations up to 0.035 mg per liter, much lower than the safe levels for daily exposure (0.400 mg per liter). Water from these sites drains into other creeks and rivers and is diluted before entering reservoirs.

Michael and others (1999) reported the dilution of hexazinone downstream of treated sites. One mile below the treated site, hexazinone concentrations were diluted to one-third to one-fifth the concentration observed on the treated site. Hexazinone was applied for site preparation at 6 pounds active ingredient (ai) per acre to clay loam soils, a rate three times the normal, and it was applied directly to

Table 13.3—Frequency and occurrence of surface and ground water contamination from pesticide use in North America[a]

Pesticide	Water type	Location	Maximum	Range	Comments	Reference
			- - - - - *Milligrams per liter* - - - - -			
2, 4-D	S	Large river basins throughout the United States	0.0075	0.00004 – 0.0075	Twenty-four reports of mainly urban, suburban, agricultural sources	Larson and others 1997
	S	Streams in Oregon and California	2.0	ND – 2.0	Highest concentrations observed from forest areas where no attempt was made to prevent application to water	Norris 1981
	G	Saskatchewan, Canada	.0000007	NG	Natural spring flow	Wood and Anthony 1997
	G	Connecticut, Iowa, Kansas, Maine, Mississippi, South Dakota	.049	.0002 – .049	Well water samples, except for South Dakota, from shallow sand and gravel aquifer	Funari and others 1995
Borax		NR	NR	NR	NR	NR
Carbaryl	S	Mississippi River	.0001	NG	One report	Larson and others 1997
	S	New Brunswick, Canada	.314	NG	Aerial spray spruce budworm control	Sundaram and Szeto 1987
	S	New Brunswick, Canada	.314	.123 – .314	Budworm control	Holmes and others 1981
Chloropicrin		NR	NR	NR	NR	NR
Chlorpyrifos	S	Mississippi River, the lower Colorado River, rivers and lakes in Kansas, irrigation ditches in California Arizona, Nevada	.00015	.00004 – .00015	Three reports	Larson and others 1997
Clopyralid		NR	NR	NR	NR	NR
Dazomet		NR	NR	NR	NR	NR
Dicamba	S	USFS land near Hebo, OR	.037	.006 – .037	Treated 166 ac of 603-ac forest catchment; highest concentration diluted to 0.006 mg/L 2.2 mi downstream	Norris 1975
Glyphosate	S	45-ha coastal British Columbian catchment	.162	.0032 – .162	Highest concentration in streams intentionally sprayed, lowest in streams with smz	Feng and others 1990
	S	Quebec, Canada	3.080	.078 – 3.08	Nine of 36 streams contained glyphosate after forest spraying	Leveille and others 1993
	S	Ohio	5.2	NG	No-tillage establishment of fescue	Edwards and others 1980
	S	Georgia Michigan Oregon	.035 1.237 .031	NG NG NG	Forest sites for scrub-hardwood control and direct spray of streams	Newton and others 1994

continued

Table 13.3—Frequency and occurrence of surface and ground water contamination from pesticide use in North America[a] (continued)

Pesticide	Water type	Location	Maximum	Range	Comments	Reference
			- - - - - *Milligrams per liter* - - - - -			
Glyphosate (cont.)	G	Newfoundland, Canada	0.045	0.004 – 0.045	Application of 4 lbs ai/ac to power substations resulted in contamination of water in monitoring wells	Smith and others 1996
Hexazinone	S	Mississippi River	.00007	NG	Detected in five tributaries	Larson and others 1997
	S	Alabama, Florida, Georgia	.037	.0013 – 0.037	Seven reports, each treated catchment containing ephemeral/first-order streams	Michael and Neary 1993
	S	Alabama	2.400	NG	Applied directly to ephemeral channel and in first runoff water	Miller and Bace 1980
	S	Alabama	.473	.422 – .473	Ephemeral/first-order stream in catchments treated with 3x rate of hexazinone in liquid and pellet formulation with accidental application to streams	Michael and others 1999
	S	Arkansas	.014	NG	11.5-ha watershed drained by ephemeral to first-order stream	Bouchard and others 1985
	S	Georgia	.442	NG	Ephemeral/first-order stream in treated catchment, pellets applied to stream channel	Neary and others 1986
	G	NG[b]	.009	NG	Only one value reported from a single study	Funari and others 1995
Imazapyr	S	Alabama	.680	.130 – .680	Two reports, each treated catchment containing ephemeral/first-order streams, herbicide accidentally applied to stream channel	Michael and Neary 1993
Methyl bromide		NR	NR	NR	NR	NR
Metsulfuron	S G	Central Florida	.008 .002	NG NG	Water in surface depression in slash pine site and 1 of 207 shallow (6-ft) well samples	Michael and others 1991
Picloram	S	North-central Arizona	.32	NG	Pinyon-juniper site	Johnsen 1980
	S	Streams and rivers in North Dakota, Wyoming, Montana	.005	.00001 – .005	Four reports from mainly rangeland uses	Larson and others 1997
	S	Alabama	.442	NG	Pellets accidentally applied directly to forest stream	Michael and others 1989
	S	Georgia, Kentucky, Tennessee	.021	ND – .021	Six study catchments with ephemeral/first-order stream in each treated forest catchment	Michael and Neary 1993
	S	North Carolina	.01	NG	Ephemeral/first-order stream in treated forest catchment	Neary and others 1985

continued

Table 13.3—Frequency and occurrence of surface and ground water contamination from pesticide use in North America[a] (continued)

Pesticide	Water type	Location	Maximum	Range	Comments	Reference
			- - - - - *Milligrams per liter* - - - - -			
Picloram (cont.)	G	Saskatchewan, Canada	0.000225	NR	Natural spring flow	Wood and Anthony 1997
	G	Iowa, Maine, Minnesota, North Dakota	.049	0.00063 – 0.049	Fewer than 2% of well samples were positive	Funari and others 1995
Strychnine		NR	NR	NR	NR	NR
Thiram		NR	NR	NR	NR	NR
Triclopyr	S	Florida	.002	NG	Coastal Plain flatwoods catchments near Gainesville, FL	Bush and others 1988
	S	Ontario, Canada	.35	.23 – .35	Intentional aerial application to boreal forest stream	Thompson and others 1991
Zinc phosphide		NR	NR	NR	NR	NR

Ai = active ingredient; G = ground water; ND = not detected; NG = not given; NR = no reports found in published literature; S = surface water; smz = streamside management zone; USFS = USDA Forest Service.
[a] This table summarizes the levels of pesticides reported in the literature at specific sites and is representative of the literature from North America. However, it cannot be extrapolated for purposes of prediction.
[b] The authors do not provide specific location.

a stream segment, resulting in a maximum observed on-site concentration of 0.473 mg per liter. This was slightly more than the lifetime HAL but considerably below the longer term HAL of 9.0 mg per liter (U.S. EPA 1990). Following the application, on-site stream concentrations did not exceed the lifetime HAL.

Norris (1975) reported contamination of streamflow with dicamba used for control of hardwoods on silty clay loam soils in Oregon. On a 603-acre watershed, 166 acres were aerially sprayed with 1 pound ai per acre [1.1 kilograms (kg) per hectare] of dicamba. A small stream segment was also sprayed causing detectable dicamba residues 2 hours after application began, approximately 0.8 miles (1.3 kilometers) downstream. Concentrations rose for approximately 5.2 hours after treatment began and reached a maximum concentration of 0.037 mg per liter, less than one-fifth of the HAL (0.200 mg per liter). No dicamba residues were detected beyond 11 days after treatment.

Glyphosate and 2,4-D have aquatic labels, which permit direct application to water. Stanley and others (1974) found that when 2,4-D was applied to reservoirs for aquatic weed control, about half of water samples from within treatment areas contained 2,4-D, and the highest concentration (0.027 mg per liter) was less than half of the HAL (0.070 mg per liter). Newton and others (1994) aerially applied glyphosate at three times the normal forestry usage rate [4 pounds ai per acre (4.4 kg per hectare)], no buffers were left, and all streams and ponds were sprayed. Initial water concentrations were 0.031 and 0.035 mg per liter in Oregon and Georgia, respectively, and 1.237 mg per liter in Michigan on the day of application. After day one, glyphosate concentrations dropped to below 0.008 mg per liter on all three sites for the duration of the study. The HAL was exceeded on only one of three sites and then for only 1 day.

There is little information on the movement of metsulfuron to streams. Michael and others (1991) found trace residues of metsulfuron in shallow monitoring wells in Florida where 24 wells were sampled to a depth of 6 feet (1.8 meters). Metsulfuron was detected (0.002 mg per liter) in 1 of 207 samples collected during 2 months after application.

Pesticides movement into streams is well documented, but movement into ground water is not well researched. During movement from streamwater into ground water, concentrations should be reduced considerably for several reasons. Infiltrating pesticides must pass through several physical barriers or layers before reaching ground water. As they pass through each layer, they are degraded, diluted, and metabolized. Surface water provides a medium for dilution, hydrolysis, and photolysis. Aquatic vegetation can metabolize pesticides. Microbes associated with coarse and fine particulate organic matter found naturally in streams also metabolize pesticides.

In order for water on the soil surface to carry pesticides into ground water, it must pass through the soil column. Here again, processes work to reduce the potential for pesticides to reach ground water. Pesticides percolating through the soil column are adsorbed to soil particles, reducing the amount reaching the ground water. Pesticides adsorbed onto soil particles may be irreversibly bound, released slowly, or further metabolized by microbes. Once pesticides reach ground water, they may degrade further. Cavalier and others (1991) found that microbes degraded herbicides, including 2,4-D, in ground water.

Thus, ground water concentrations of pesticides should be considerably lower than observed in surface water. Funari and others (1995) reviewed the literature and reported the range of maximum ground water concentrations of pesticides, including those used in forestry, agriculture, home and garden, and on industrial rights-of-way. The maximum range of values for 2,4-D (0.0002 to 0.0495 mg per liter), hexazinone (0.009 mg per liter), and picloram (0.00063 to 0.049 mg per liter) are much lower than the HAL's for those compounds.

The National Water-Quality Assessment Program (NAWQA) conducted by the U.S. Geological Survey began in 1991. The focus of NAWQA is to identify nutrient and pesticide contamination of water throughout the United States. The 1999 NAWQA report (found at http://water.usgs.gov/pubs/circ/circ1225/index.html) makes little mention of forest sites or forestry pesticides, but concludes that: "Concentrations of nutrients and pesticides in streams and shallow groundwater generally increase with increasing amounts of agricultural and urban land in a watershed." The report focused on more than 50 major river basins and aquifers supplying water to more than 60 percent of the population and approximately half of the area of the United States. Few forestry pesticides other than 2,4-D were found in these basins or aquifers.

Even in predominantly agriculture areas, the report states:

> One of the most striking results for shallow groundwater in agricultural areas, compared with streams, is the low rate of detection for several high-use herbicides other than atrazine. This is probably because these herbicides break down faster in the natural environment compared to atrazine.

Atrazine is principally used in growing corn (maze). It has not been used on NFS land since 1992. While not directly addressing forestry pesticides and drinking water, these NAWQA conclusions support the above research findings and conclusions that ground water contamination by pesticides should be lower than observed for surface water. Because surface water contamination from forest sites treated according to label directions does not exceed HAL's, it is very unlikely that ground water contamination would exceed HAL's.

Several of the pesticides in table 13.2 have not been reported in water. They include chloropicrin, chlopyralid, dazomet, and thiram. Chloropicrin and dazomet are soil fumigants, which are gases in their active form and are used only for seedling production. Chlopyralid is a relatively new compound in the United States. Thiram is a dimethyl dithiocarbamate fungicide, principally used in forestry for seed protection.

There is very little water-quality data for pesticides used in nursery disease control and soil fumigation. More than 71 percent of fungicides and fumigants used on NFS land are applied in nurseries. Intense use in a nursery may result in localized ground water contamination. Three pesticides (chloropicrin, dazomet, and methyl bromide) make up this group of intensively used agents. Chloropicrin is toxic to plants and is used in combination with other chemicals for fumigating seedbeds. Dazomet, a soil fumigant, is a gas and is relatively insoluble in water (3 grams per liter). However, dazomet is unstable in water and quickly breaks down into methyl isothiocyanate (MITC), formaldehyde, monomethylamine, and hydrogen sulfide. All are toxic, but the most toxic is MITC. The RfD for formaldehyde is 0.2 mg per kilogram per day. However, EPA has classified formaldehyde as a compound of medium carcinogenic hazard to humans. Methyl bromide is very toxic. Data are insufficient to determine whether frequent use of these three pesticides adversely impacts water quality, either locally or over an expanded area.

Reliability and Limitation of Findings

Most data reviewed in this chapter come from scientific literature. The data listed in table 13.3 and derived from Larson and others (1997) were extracted from in-house reports from the U.S. Geological Survey, the EPA, State, and local governmental departments for the environment and scientific literature. Reports published in scientific literature are the most reliable because they were subject to peer review and scrutiny for validity of methods, completeness of data, and interpretation of the data. Monitoring data from in-house publications and reports may be less reliable.

Some variability in results of individual studies is due to regional soil and climate differences. In the South, infiltration rates on many forestry sites are generally low, owing to the highly eroded condition of the soils. Here, precipitation intensity frequently exceeds infiltration rates, producing overland flow on newly prepared sites. Overland flow may lead to much higher pesticide concentrations in stormflow than in other areas of the country with much higher infiltration rates. Very high infiltration rates are typical of soils in the Pacific Northwest. Therefore, if streams are protected by buffers, broadcast application of pesticides generally results in stream contamination either via direct application or through baseflow contributions. In general, levels of contamination are lowest where infiltration rates are highest.

Care must always be exercised in extrapolating data from local studies on drinking water to a regional or larger scale. However, three strategies of worst-case scenarios used in these studies mitigate against high levels of uncertainty: (1) several studies have investigated the impacts of pesticides applied directly to surface water; (2) several studies have investigated the impacts on water of pesticides applied at several times the prescribed rate; and (3) most of the studies conducted specifically on forestry sites treated the entire catchment from which water samples were taken, resulting in samples with levels of pesticide contamination greater than are likely to occur anywhere downstream.

Research on the impacts of pesticides applied directly to surface water used the worst-case scenario for forest operational treatments in which pesticide was applied at normal rates directly to surface water (ponds and streams). These studies did not find any contamination of water at levels above the HAL for any pesticide studied. Research on aquatic impacts from pesticides applied at several times the labeled rate used the worst-case scenario for operational treatments where an area might receive multiple applications in error or where small spills occurred. In these studies, HAL's were exceeded by only a few percent and only briefly, usually for less than a few hours. Both worst-case scenarios just described were combined with the third worst-case scenario in which all sampling was conducted on surface water found within the treated area. In this case, most of the water was in small pools or ephemeral to first-order streams. While ephemeral to first-order streams or pools are unlikely to be drinking water sources because of low yield, they do represent water most likely to be severely contaminated from normal forest pesticide applications. Even these waters were not contaminated at levels exceeding HAL's, except where pesticide was applied at several times the labeled rate as described above.

In addition, data on contamination of water for the pesticides in table 13.3 have been taken from a number of studies conducted in North America and the findings are generally similar. These studies have, with a few exceptions, confirmed the absence of significant contamination of drinking water. The exceptions were those cases in which a pesticide was applied directly to water, and the high concentrations observed in those studies were at or only slightly above drinking water standards. These high concentrations lasted only a few hours at most before dropping well below current HAL's. It is clear from the available literature that use of pesticides in strict accordance with label directions on NFS land cannot be expected to contribute significantly to ground water or drinking water contamination. It is also clear that pesticides, unless clearly labeled for aquatic uses, must not be applied directly to water, and that pesticides should be used around water resources, which are particularly sensitive only after careful consideration of the ramifications.

Limitations of the data are obvious for the few chemicals that have not been investigated. We need data on them as well as other chemicals about which little information is available. Additional limitations include lack of sufficient testing for health effects as indicated in table 13.2. The question of cumulative toxicological effects has not been addressed for any of the pesticide mixtures utilized in modern forest management.

Research Needs

Several issues related to vegetation management need additional research.

1. One issue is impacts of frequent, repeated use of fungicides and fumigants in nursery operations on nearby water quality.
2. Another issue is effectiveness of buffer width and composition. There is too little information on the processes and interactions of site-specific characteristics with pesticide chemistry that permit buffers to mediate against contamination of streams and surface waters in general. These processes and the interactions of pesticide chemistry with site-specific conditions must be identified and understood so managers can design and install optimally functional buffers to protect the water resource and its associated aquatic ecosystem.

3. A third issue is that several pesticides in table 13.3 lack published reports relating their use to occurrence in water.
4. Still another issue is the effects of commonly used pesticide mixtures, as opposed to single compounds, on the water resource.

Key Points

Relative to agricultural, urban, and other uses of pesticides, very small amounts are used on NFS land. Further, the use patterns for any specific piece of land are infrequent, except in the case of vegetation protection from pathogens, animals, and insects where annual treatment may be required, especially in greenhouses and nurseries. Pesticides used on NFS land are also used around the World to accomplish management goals similar to those on NFS land, but often in a much more intensive way. Even with the widespread use of pesticides in North America, those typically used on NFS land have not been identified in surface or ground water at sufficiently high concentrations as to cause drinking water problems. Their rapid break down by physical, chemical, and biological routes coupled with use patterns precludes the development of water contamination problems unless they are applied directly to water. Even though these same pesticides are used around homes, in urban and in agricultural settings, their use in forest management is still controversial in the public arena. Therefore, their use should be carefully planned and all agency, local, State, and Federal laws should be followed. It is especially important to follow all label directions because pesticide labels are legal documents specifying Federal laws pertaining to their use. Best management practices should be carefully adhered to and use around drinking water supplies should be avoided, except where permitted by the label. Wherever pesticides are used, precautions should always be taken to protect drinking water sources from contamination.

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