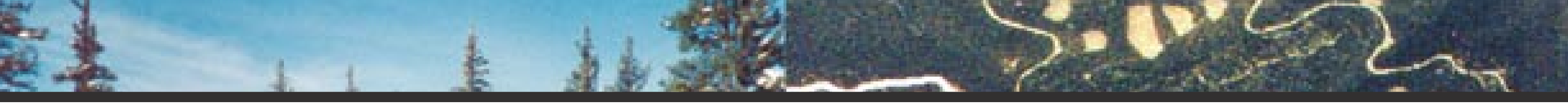

Lee H. MacDonald and John D. Stednick

Forests and Water:
A State-of-the-Art Review for Colorado

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CWRRI Completion Report No. 196

Colorado State University

Knowledge to Go Places

Cover photos from upper left in clockwise order:

The winter snowpack in the sub-alpine and alpine zones produces most of Colorado's runoff.

Aerial photo of a portion of the Coon Creek experimental watershed in south-central Wyoming. The forest cover was removed from 24% of this 4130-acre watershed by cutting 240 openings 3 to 10 acres in size and building the associated road network. A primary purpose of the study was to determine whether the flow increases observed on small experimental watersheds could be scaled up to larger basins.

View of the North Fork of the South Platte River southeast of Pine, Colorado. The higher portions of the South Platte River basin provide much of the water for the Denver metropolitan area and other communities to the north and east.

View of a small swale that burned at high severity in June 2002 Hayman fire. Prior to the fire there was 80% ground cover and no distinguishable channel. Picture was taken after a 0.67-inch rain event in July 2002.

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Colorado State University

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2003

CWRRI Completion Report No. 196

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Additional copies of this report can be obtained from the Colorado Water Resources Research Institute, E-102 Engineering Building, Colorado State University, Fort Collins, CO 80523-2033 (970-491-6308 or e-mail: cwrri@colostate.edu), or downloaded as a PDF file from http://cwrri.colostate.edu.

Executive Summary

1. Forests occupy 22.6 million acres in Colorado, or 32% of the land area. Nearly three-quarters of the forest lands in Colorado are in public ownership. About 55% of the forested area is considered as suitable for forest harvest. National forests comprise nearly half of the forested area and approximately 60% of the area considered suitable for forest harvest. There are no significant, privately-owned, industrial forest lands in Colorado.

2. Most of the runoff in Colorado comes from forest and alpine areas above approximately 9000 ft (2730 m) in elevation, and is generated by snowmelt from approximately mid- April to mid-July. The dominant role of high-elevation areas is due to the increase in precipitation with increasing elevation, the decrease in potential evapotranspiration with increasing elevation, and the concentration of snowmelt in a relatively short period of time. In lower elevation forests the amount of runoff per unit area is greatly reduced because the rainfall and snowmelt inputs are much smaller relative to potential evapotranspiration.

3. Research on the relationships between forests and water has been conducted in Colorado since 1910. Intensive studies from Wagon Wheel Gap, the Fraser Experimental Forest, Manitou Experimental Forest, and other sites provide a thorough understanding of how changes in forest cover affect evapotranspiration, soil moisture storage, and the amount and timing of runoff. This knowledge base is strongest for the higher-elevation spruce-fir and lodgepole pine forests, as these forest types generate most of the runoff from forested areas and have been studied more intensively.

4. Annual water yields in the higher elevation spruce-fir and lodgepole pine forests are inversely proportional to the amount of forest canopy as indexed by basal area. Complete removal of the forest canopy in the sub-alpine zone can increase annual water yields by as much as 8 inches (20 cm) of water per unit area. This increase in water yield is due to the reduction in winter interception losses and summer evapotranspiration. Nearly all of this additional runoff comes on the rising limb of the snowmelt hydrograph (i.e., early May to mid-June), and the increases in runoff are several times larger in wet years than dry years. This implies that water storage facilities are required if an increase in runoff is to be carried over into the summer or from year to year.

5. A reduction in the amount of forest canopy will increase the rate of spring snowmelt, and complete removal of the forest canopy will increase the size of the annual maximum peak flows by approximately 40-50%. Paired-watershed studies have shown that removal of the forest canopy has no significant effect on summer low flows, as the water "saved" by reducing summer evapotranspiration is simply carried over to the following spring.

6. Reducing forest density has a progressively smaller effect on annual water yields with decreasing annual precipitation. Both paired-watershed studies and plot-scale research show that reducing forest density has no detectable effect on water yields when annual precipitation is less than 18-19 inches or approximately 460 mm. Paired-watershed studies also indicate that at least 15% of the forest canopy within a watershed must be removed in order to obtain a measurable increase in annual water yields from small research watersheds. The detection of change becomes much more difficult in larger watersheds when discharge is being measured with standard techniques in natural channels.

7. The increase in water yield that results from timber harvest or other disturbances will decrease as the forest regrows. Paired watershed studies suggest that it will take approximately 60 years until annual water yields return to their pre-disturbance levels in the higher-elevation spruce-fir and lodgepole pine forests. Hydrologic recovery is substantially faster in aspen forests due to faster regrowth and in drier forest types.

8. Historic photographs, forest stand records, and other data indicate that forest density in Colorado is generally greater than in the mid to late 1800s. This increase in forest density is attributed to suppression of forest fires, reduced grazing, and lower rates of forest harvest for timber, fuel, and other products. The human-induced changes in forest density and composition are most pronounced in the low and mid-elevation ponderosa pine and mixed conifer forests, as these forests were regularly subjected to both low-intensity surface fires and high-intensity, stand-replacing

fires. At higher elevations the changes in forest composition and density are not as apparent, primarily because the duration of the fire suppression period is short relative to the natural fire recurrence interval.

9. The changes in forest density and cover over the last century or so are generally believed to have decreased annual water yields. Annual water yields from the 1.34 million acres of national forest lands in the North Platte River basin are estimated to have decreased by approximately 11 to 13 percent or 150,000 to 190,000 acre-feet per year, depending on the assumed stand history for the spruce-fir forests. Hydrologic models indicate that average annual water yields could be increased in the North Platte River basin by about 55,000 acre-feet per year if all 502,000 acres designated as suitable for timber harvest was regularly cut on a sustained yield basis. Similar data are not available for other river basins in Colorado, although the overall trends are probably similar.

10. The rates of timber harvest on national forest lands in Colorado substantially increased in the early 1960s, peaked in the late 1960s and late 1980s, and then dropped by about two-thirds in the 1990s relative to peak harvest levels. The projected increase in annual water yields due to timber harvest on national forest lands was nearly 100,000 acre-feet in the older forest management plans, but is projected at 30,000 to 40,000 acre-ft per year in the most recent forest management plans. Recent harvest levels have averaged slightly more than half of the values in the selected alternative in the most recent forest plans. Fuels treatments are projected to increase average annual water yields by 18,000 acre-feet/yr for the eight national forests that estimated these values in their forest management plans, and most forests met their acreage targets for fuels treatments from 1997 to 2000. Because the average annual runoff from national forest lands in Colorado is greater than 10 million acre-feet per year, the projected water yield increases from forest harvest and fuels treatments on national forest lands represent less than 1% of the total water yield.

11. The quality of the water flowing from forested areas is generally very high. Forest harvest and fuels treatments should have minimal adverse effects on water quality if they are carefully designed and conducted in accordance with best management practices. Wildfires pose the biggest threat to water quality and site productivity. In severely-burned areas peak runoff rates can increase by a factor of 10 of more, while erosion rates can increase by 100 times relative to unburned areas. These large increases are due to both the lack of cover and the development of a water-repellent layer at or just below the soil surface. Burning at moderate and low severities causes much smaller increases in the size of peak flows and erosion rates. Data from the Front Range suggest that there is little recovery by the second summer after burning, but the higher runoff and erosion rates should decline to near-background levels by 3-4 years after burning. Percent bare soil appears to be the dominant control on post-fire erosion rates, so treatments that immediately increase the amount of ground cover are most likely to be effective in reducing post-fire erosion rates.

12. This report does not attempt to address the myriad of other issues that must be considered when evaluating various management alternatives for forested lands. Some of these issues include the numerous laws and regulations that affect land management, economic considerations, the downstream uses of water and water storage capacities, and the effects of forest management on recreation, local communities, aesthetics, and other plant and animal species.

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Dr. MacDonald is professor of land use hydrology in the Watershed Science Program at Colorado State University. He takes a process-based approach to analyzing the effects of land use on runoff and erosion, and how these changes affect streams and wetlands. Dr. MacDonald has a broad academic background, including a B.S. in human biology from Stanford University, a M.S. in resource ecology from the University of Michigan, and a Ph.D. in forest hydrology from the University of California at Berkeley. At CSU he has published six major monographs and more than 40 scientific articles on a variety of land use issues, including how forest management activities affect runoff and erosion; hillslope and wetland hydrology; effects of wild and prescribed fires on runoff and erosion; cumulative watershed effects; road erosion; and monitoring the effects of land management activities on streams. His diverse work experience includes setting up research and training programs in developing countries for the United Nations University, working as a hydrologist for the U.S. Forest Service, consulting for a wide variety of public and private organizations, and a one-year post-doctoral position at the Center for Streamside Studies at the University of Washington.

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The publication of this report is the result of contributions from a large number of people and agencies. We would particularly like to thank Dr. Robert Ward and the Advisory Board of the Colorado Water Resources Research Institute, as they initiated this project and contributed a substantial part of the overall budget. Additional contributions from the Colorado River Conservation Board, Denver Water Board, and Northern Colorado Water Conservancy District helped make this project possible, and we would like to respectively thank Rocky Wiley, Dave Merritt, and Brad Wind for their guidance and reviews of earlier drafts.

Chuck Troendle generously contributed his knowledge and experience, and we are very grateful for his willing participation throughout the extended writing and review process. Polly Hays and Carl Chambers of the USDA Forest Service participated in several meetings and provided many helpful comments. Other reviewers included Don Anderson, Greg Bevenger, Dan Binkley, Joan Carlson, Frank Cross, Les Dobson, Dave Gloss, Ron Gosnell, Kent Holsinger, Forrest Leaf, Jim Meiman, Phil Omi, Bill Romme, Larry Schmidt, Liz Schnackenberg, and Andrea Holland Sears, and we thank them for their assistance in correcting and improving the structure and content of this report. A number of graduate students assisted in the preparation of this report, and we are grateful to Drew Coe, Tedd Huffman, Zamir Libohova, Sam Litschert, and Steve Thomas.

Our goal was to make this report as accurate and complete as possible, and this required us to cover a surprisingly wide range of topics. While some may argue that it is too detailed, we felt the need to present the complexity of the relationships between forests and water. Any oversimplifications or misrepresentations are solely the responsibility of the authors, but we are confident that the report is a sound summary of the overall state of knowledge. We hope that the information presented in this report will provide the platform for future discussions of how we might manage our forest resources, and the likely consequences of different management scenarios. If the report ultimately leads to more informed decisions, the time and funds invested by all of the participants will have been well spent.

Chapter 1
Introduction and Objectives

1.1. Introduction and Background

The relationship between forests and water is an important issue in Colorado because forests occupy a large percentage of the land area and forested areas supply much of the state's water. Forests and woodlands cover approximately 35,300 mi^2 within Colorado, or approximately one-third of the state (DNR, 2002). In order of prevalence, the dominant forest types are pinyon-juniper (33%), spruce-fir (21%), ponderosa pine (15%), aspen (14%), lodgepole pine (10%), Douglas fir (5%), and other forest types (3%) (Figure 1.1).

Much of Colorado is classified as semi-arid and at lower elevations only a small proportion of the annual precipitation is converted into runoff. Annual precipitation increases rapidly with elevation (Figure 1.2), and there is an even more rapid increase in the amount of runoff per unit area due to the associated decrease in potential evapotranspiration (Figure 1.3).

The net result is that the higher-elevation forests (e.g., above about 9,000 feet) generate much of the state's water supply. The amount, timing, and quality of this runoff is a critical issue to the citizens of Colorado, private companies, and government agencies. Natural and human-induced disturbances can greatly affect the amount and type of forest cover, and this has direct implications for the amount and quality of runoff.

The effects of forest management on the quantity, quality, and timing of runoff have long been of interest to land managers and water users, and a topic of considerable debate. The ancient Greeks noted that clearing the forests could cause springs to dry up (Biswas, 1970), while a large number of recent studies have shown that forest clearing can increase annual water yields (reviews by Bosch and Hewlett, 1982; Stednick, 1996). The large variation in the hydrologic effects of forest management means that one or more studies can be found to support nearly any point of view. The resolution of these appar-

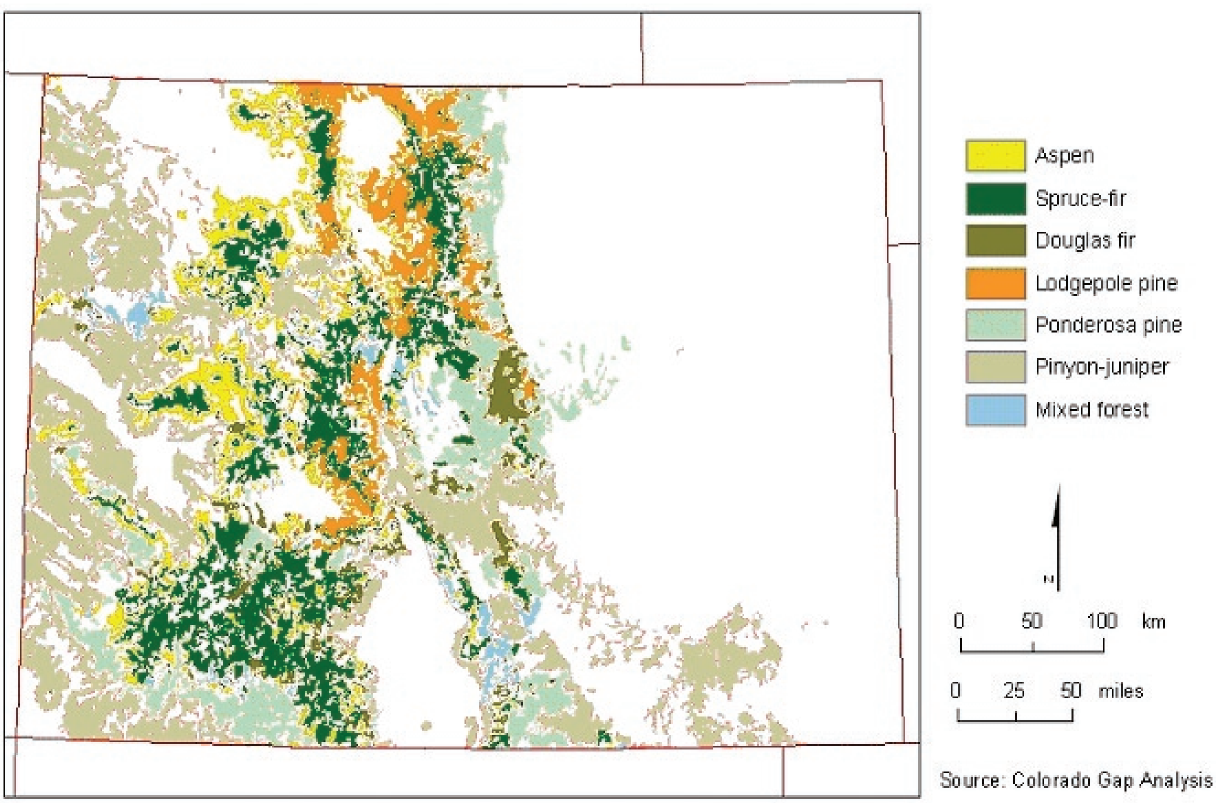


Figure 1.1. Map of the major forest types in Colorado.

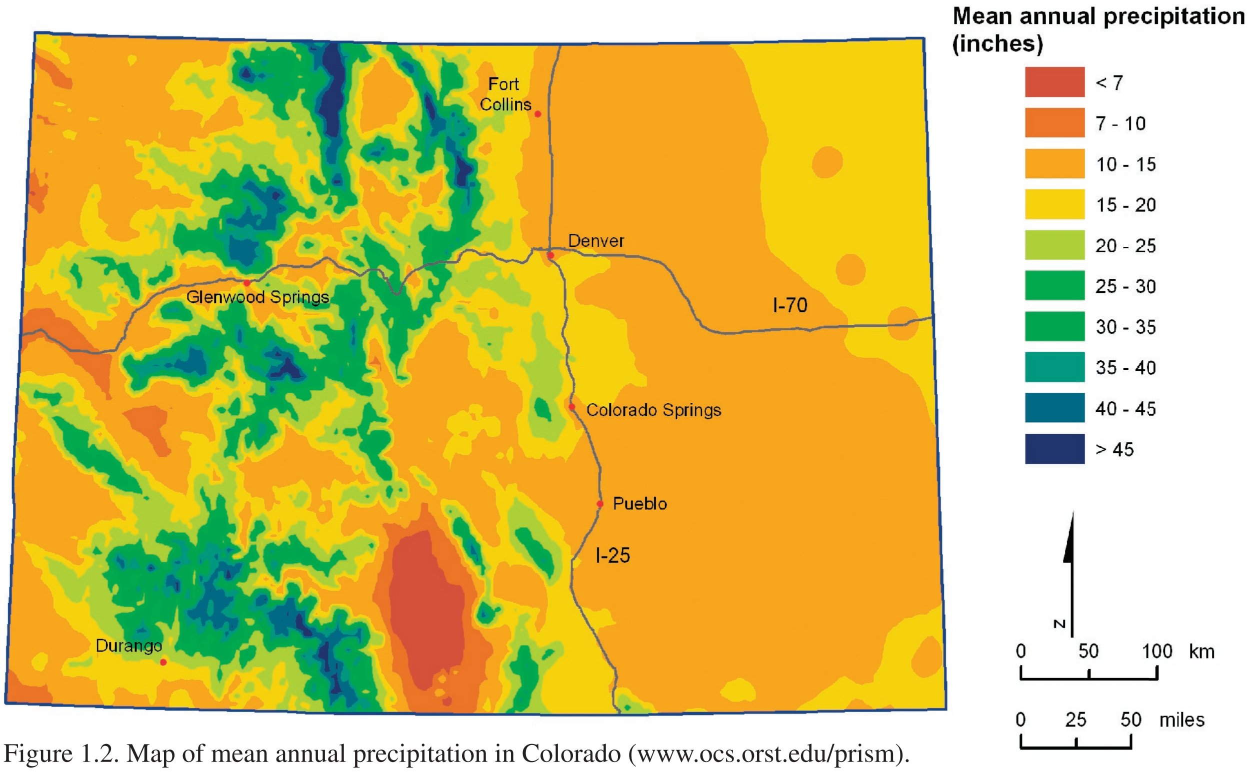


Figure 1.2. Map of mean annual precipitation in Colorado (www.ocs.orst.edu/prism).

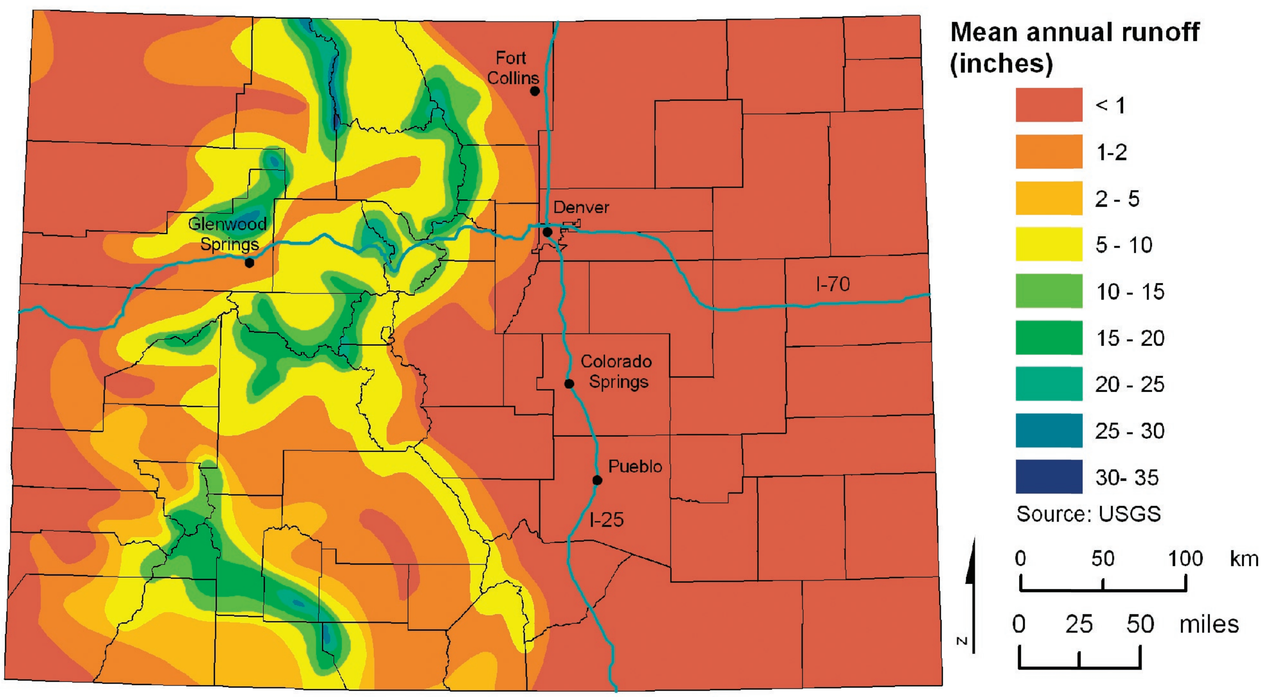


Figure 1.3. Map of mean annual runoff in Colorado.

ently disparate results requires a basic understanding of forest hydrology, including infiltration, interception, soil moisture storage, and evapotranspiration. Different types of forest disturbance and forest regrowth can affect each of these processes in different ways, depending on the site conditions, how a given management action is carried out, and the hydrologic event of concern (e.g., annual water yields, summer low flows, spring snowmelt, or extreme rain events). Generalizations and extrapolations can be misleading unless there is a clear linkage to, and understanding of, the underlying processes.

There is a close relationship between forest condition, runoff processes, and the quality of the water emanating from a forest. In general, forests grow in more mesic areas and generate relatively little overland flow at the hillslope scale (Dunne and Leopold, 1978). As a result, forested areas commonly provide large amounts of high-quality water (Dissmeyer, 2000). Poor forest management practices can degrade the quality of this water, while careful forest management can alter the amount of runoff with little or no detectable effect on water quality.

These generalizations apply to Colorado, as most of the state's runoff emanates from forested areas at higher elevations, and this water is generally of very high quality. Rapid population growth is increasing the demand for water in urban areas, and this increased demand is illustrated by the rapid increases in the price of water along the Front Range. Increasing water demands are also emanating from industry, recreational interests, and the desire to restore natural flow regimes to improve aquatic habitat and support endangered species. Municipal water suppliers are required to assess the risks to their sources of water, and in most cases this means an assessment of how forest conditions affect water quality. Substantial amounts of public and private funds are being spent to improve water quality and aquatic habitat. Given these demands and issues, there is an increasing need for a more rigorous understanding of how current forest conditions and possible management actions can affect the amount, timing, and quality of runoff.

Current forest conditions, water yields, and water quality also need to be evaluated in a broader historical context, as past changes in land use and forest composition are directly and indirectly affecting the amount of runoff as well as the quality of that runoff. For example, fire suppression and low levels of forest harvest are believed to have increased the density of forest vegetation and reduced annual water yields in the North Platte watershed (Leaf, 2000; Troendle and Nankervis, 2000). The increase in forest density can indirectly affect water quality, as denser forests tend to be at greater risk to high-severity wildfires. Large, high-severity wildfires are of particular concern because they can greatly increase the size of peak flows, hillslope erosion rates, and downstream sedimentation rates (Tiedemann et al., 1979; Robichaud et al., 2000). The effects of high-severity fires on runoff and water quality were dramatically illustrated by the 1996 Buffalo Creek fire, where post-fire flooding and erosion led to the loss of human life, severe property damage, and greatly increased water treatment costs (Agnew et al., 1997; Moody and Martin, 2001). The large wildfires in 2002 focused public attention on the changing wildfire risk due to prolonged drought and high forest densities.

From a regulatory perspective, a detailed understanding of the relationships between forests and water are needed to meet the demands of several important pieces of environmental legislation. The National Environmental Policy Act requires federal agencies to assess the likely impact of proposed federal actions, including a larger-scale assessment of potential cumulative impacts (CEQ, 1997). Regulations emanating from the Clean Water Act and its amendments require a watershed approach to improve water quality when water quality standards are not being met despite the application of NPDES permits for point sources and Best Management Practices (BMPs) for nonpoint sources (EPA, 1991). The Endangered Species Act (ESA) requires federal agencies to protect endangered species and their habitat, and many of the species of greatest concern are aquatic species that depend on the water in, or emanating from, forested areas (MacDonald, 2000). The net result is a renewed interest and focusing of public attention on the interactions between forest management and the amount, timing, and quality of runoff.

1.2. Objectives and Organization

The basic purpose of this report is to provide a state-of-the-art summary on how forest management in Colorado affects water quantity and quality, and to identify key gaps in knowledge. This synthesis should help guide public policy and decision-making, and help identify future research priorities.

The preparation of this report was conducted under the guidance of a panel of water managers and scientists. This panel was first convened in April 2000, and the specific objectives were defined as:

1. Identify the forest management issues of primary concern for water management in Colorado;
2. Summarize the state-of-the-art with respect to how past and present forest management is affecting annual water yields, low flows, and peak flows;
3. Determine the potential for altering current streamflow regimes through forest management;
4. Summarize the state-of-the-art with respect to the effects of forest management on water quality;
5. Assess the different risks to water quality posed by forest management activities, including the no action alternative;
6. Recommend areas where additional research is needed and feasible.

The following chapters sequentially address each of these objectives. Chapter 2 reviews how changes in forest cover affect annual water yields, the size of peak flows, and the size of low flows, respectively. Chapter 3 is a synthesis of the effects of forest management on water quality and sediment yields. Chapter 4 summarizes what is known about the historic changes in forest vegetation in Colorado and how these changes are affecting the amount of runoff. Similarly, Chapter 5 assesses the likely effects of past and future forest management activities on water quality. Chapter 6 uses the information from Chapters 2-5 to identify current gaps in knowledge and suggest priorities for future research.

Chapter 2
Effects of Forest Management on Runoff

2.1. Introduction

Colorado is relatively fortunate with respect to the amount of scientific research that has been conducted on the relationships between forest management and runoff. The first paired-watershed experiment in the world was conducted at Wagon Wheel Gap in southern Colorado in the early 1900s, and this was followed by a long series of studies designed to more rigorously document the effects of forest harvest on annual water yields, the size of peak flows, low flows, and, to a lesser extent, water quality. Many of the most relevant studies have been conducted on the Fraser Experimental Forest (FEF) northwest of Winter Park, Colorado (Figure 2.1) (Alexander et al., 1985; Alexander, 1987). This was established in 1937, and for the last 60 years the FEF has been the pre-eminent center for forest hydrology research in the central Rocky Mountains. Similarly, the Manitou Experimental Forest has been the site of numerous plot and watershed-scale studies in the ponderosa pine and mixed conifer zone (Figure 2) (e.g., Berndt, 1960; Gary, 1975). Information relevant to Colorado can also be drawn from watershed studies in states with similar conditions (e.g., Arizona, Idaho, Wyoming, and Utah), and a much broader pool of forest hydrology research conducted in other areas. Information from all of these sources is used in this report. The first three sections of this chapter summarize existing knowledge on the effects of vegetation management on annual water yields, the size of peak flows, and the size of low flows. The next two sections of this chapter respectively review the effects of roads and fires on runoff. The final section in this chapter discusses the extent to which results from plot or small watershed studies can be extrapolated to larger areas, and the magnitude of changes in runoff that might be expected in large watersheds. Since forest harvest is the primary technique used to investigate the relationship between forests and water, this necessarily is the primary focus of this chapter. The removal of forest cover by insects, disease, or windthrow generally should have a similar effect, but the difficulty of conducting controlled experiments and the paucity of data precludes a detailed, quantitative review of the effects of these other types of disturbance on runoff and water quality. Similarly, there

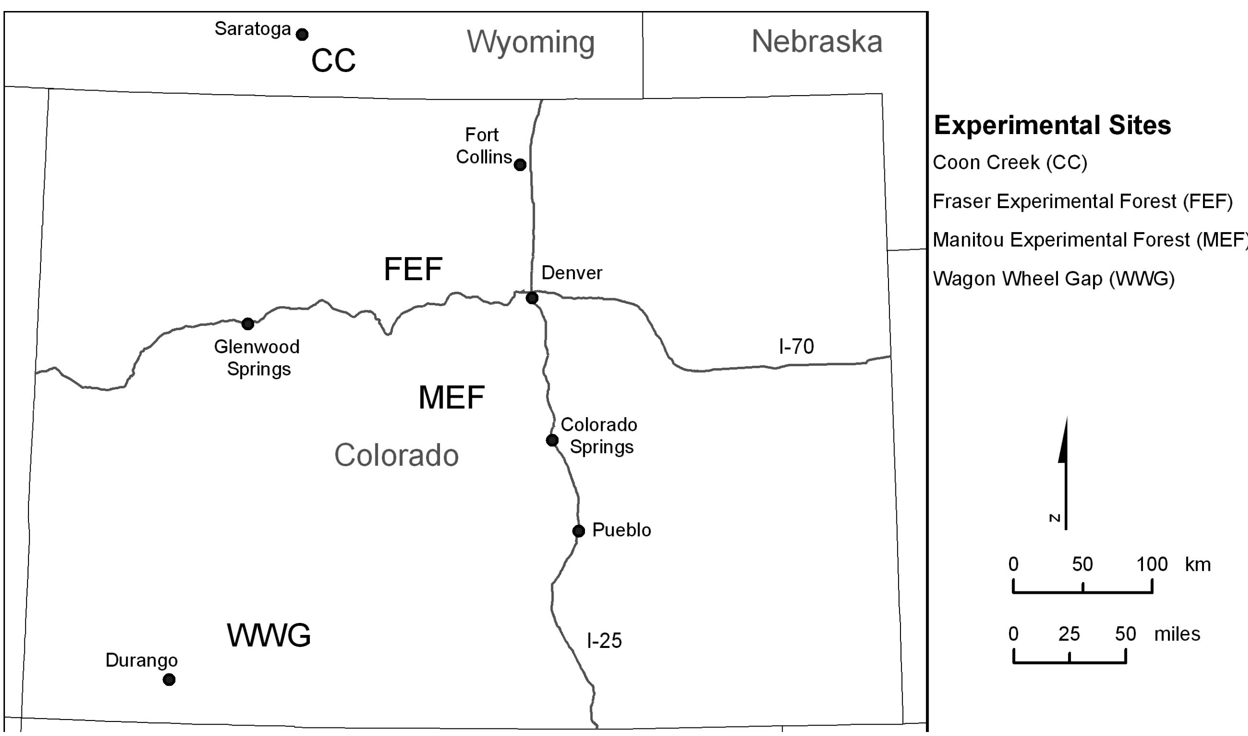


Figure 2.1. Location of experimental forests in Colorado and selected paired watershed experiments.

are relatively few well-controlled studies on the effects of fires on runoff, but the combination of process-based studies and field observations provides a reasonable basis for understanding and prediction.

2.2. Effects of Changes in the Forest Canopy on Runoff

2.2.1. Overall water balance

A useful starting point for understanding the effects of forest management on runoff is the water balance equation. In simplest terms:

$$\text{Runoff} = \text{Precipitation} - \text{Evaporation} - \text{Transpiration} + \text{Change in Storage} \quad (1)$$

In many instances the evaporation and transpiration terms are combined into a single evapotranspiration (ET) term, as in practice it is very difficult to distinguish the water being lost to the atmosphere by evaporation from the water being lost to the atmosphere through the stomata of plants by transpiration. However, it is useful to separate these two components, as changes in vegetation type and density can have different effects on these two processes under different conditions. It is also important to note that evaporation includes the loss of water by interception, which is the evaporation or sublimation of water captured on plant surfaces during or immediately after a precipitation event (Dunne and Leopold, 1978).

In many forested areas the change in storage can be ignored because the amount of water stored as soil and ground water shows little change from one year to the next (Hewlett, 1982). This assumption is generally valid for Colorado, as the amount of stored water in undisturbed forests generally reaches a similar minimum value at the end of each summer dry season (Troendle and Meiman, 1986). Changes in the amount of stored water are often significant over shorter time scales (e.g., seasonal, monthly), but on an annual time scale the change in storage in undisturbed forests is usually small relative to the errors in the other terms in equation 1.

If we assume constant precipitation from year to year, equation 1 indicates that runoff will be directly proportional to the amount of evaporation and transpiration. It follows that a reduction in vegetation cover will reduce the amount of interception and transpiration. If there is not a corresponding increase in the amount of evaporation from the soil, annual runoff must increase. In relatively dry areas or in dry years a reduction in vegetation has little or no effect on runoff because the water that is "saved" by the reductions in interception and transpira-

Table 2.1. Summary of data from paired-watershed experiments in Colorado and northern Arizona, including pre- and post-treatment water yields, elevation, and percent of vegetation removed by forest harvest.

Watershed	Area (mi²)	Elevation (feet)	Vegetation type	Veg. removed (%)	Mean annual precipitation (inches)	Mean annual runoff (inches)	Initial increase in runoff (inches)	Source
Wagon Wheel Gap, south-central Colorado		9,300	Spruce	100	21	6.1	1.1	Bates and Henry, 1928
Fool Creek, Fraser Experimental Forest (FEF)		9,600	Spruce-fir, lodgepole pine	50	30	8.7	3.2	Troendle and King, 1985
Deadhorse Creek, FEF Upper Basin	0.3	9,400-11,600	Spruce-fir, lodgepole pine	30	NA	22.5	3.6	Troendle and King, 1987
North Fork				36	32.3	15	2.4	Troendle and King, 1987
Coon Creek, south-central Wyoming	6.6	8,800-11,000	Lodgepole pine, spruce-fir	24	34.3	17.4	3.0	Troendle et al., 2001
Beaver Creek, northern Arizona		5,600-8,500	Ponderosa pine		21.7-31			
Watershed 12	0.7	7,100	Ponderosa pine	100	24.3	5.9	2.4	Baker, 1986
Watershed 17	0.5	6,900	Ponderosa pine	77	28.6	8.1	2.5	Baker, 1986
Watershed 8	2.8	7,300	Ponderosa pine	33	27.4	6.7	2.9	Baker, 1986
Watershed 16	0.4	7,100	Ponderosa pine	68	27.7	5.3	2.8	Baker, 1986
Watershed 14	2.1	7,200	Ponderosa pine	57	25.6	4.6	1.3	Baker, 1986
Watershed 9	1.8	7,200	Ponderosa pine	31	25.4	6.1	1.0	Baker, 1986
Workman Creek, central Arizona	0.5	6,600-7,800	Douglas-fir, Ponderosa pine	79	32.8	3.3	0.9	Hibbert and Gottfried, 1987

tion are simply lost to increased evaporation from the soil (e.g., Bates and Henry, 1928; Troendle, 1987a). (The word “saved” is in quotation marks because on a global scale water is generally not created or lost, and the concept of “saved” water is only valid for the area under discussion.) A 1982 review of paired watershed experiments noted that annual precipitation must exceed 450-500 mm (18-20 inches) in order to detect an increase in runoff as a result of removing much of the vegetative cover (Figure 2.2) (Bosch and Hewlett, 1982).

As annual precipitation increases beyond 450-500 mm, vegetation density increases and there is a corresponding shift from soil evaporation to interception and transpiration as the dominant sources of ET or water “loss.” ET can be predicted for the higher elevation forests in the Fraser Experimental Forests by equation 2:

$$ET = 460 \text{ mm} + 0.28\ (P - 460 \text{ mm}) \qquad (2)$$

where P is the annual precipitation in millimeters (Troendle and Reuss, 1993). This equation indicates that until annual precipitation exceeds 460 mm (18 inches), all of the precipitation is used for ET. Twenty-eight percent of all precipitation beyond this threshold will be lost to interception, and the balance becomes runoff.

These principles mean that vegetation removal will result in progressively larger increases in annual runoff as precipitation increases. Reviews of paired catchment experiments in Colorado, the U.S. (Stednick, 1996), and throughout the world (Bosch and Hewlett, 1982) show that this general trend is maintained up to at least 1600 mm (approximately 60 inches) of annual precipitation (Figure 2.2). Table 2.1 summarizes the initial increases in annual water yields that have been observed from paired-watershed studies in or particularly relevant to Colorado. The observed increases in annual water yields range from approximately one inch at Wagon Wheel Gap to 3.6 inches in the Upper Basin at Deadhorse Creek in the FEF. Larger increases are generally associated with greater amounts of precipitation, higher pre-treatment water yields, and removal of a greater proportion of the forest cover.

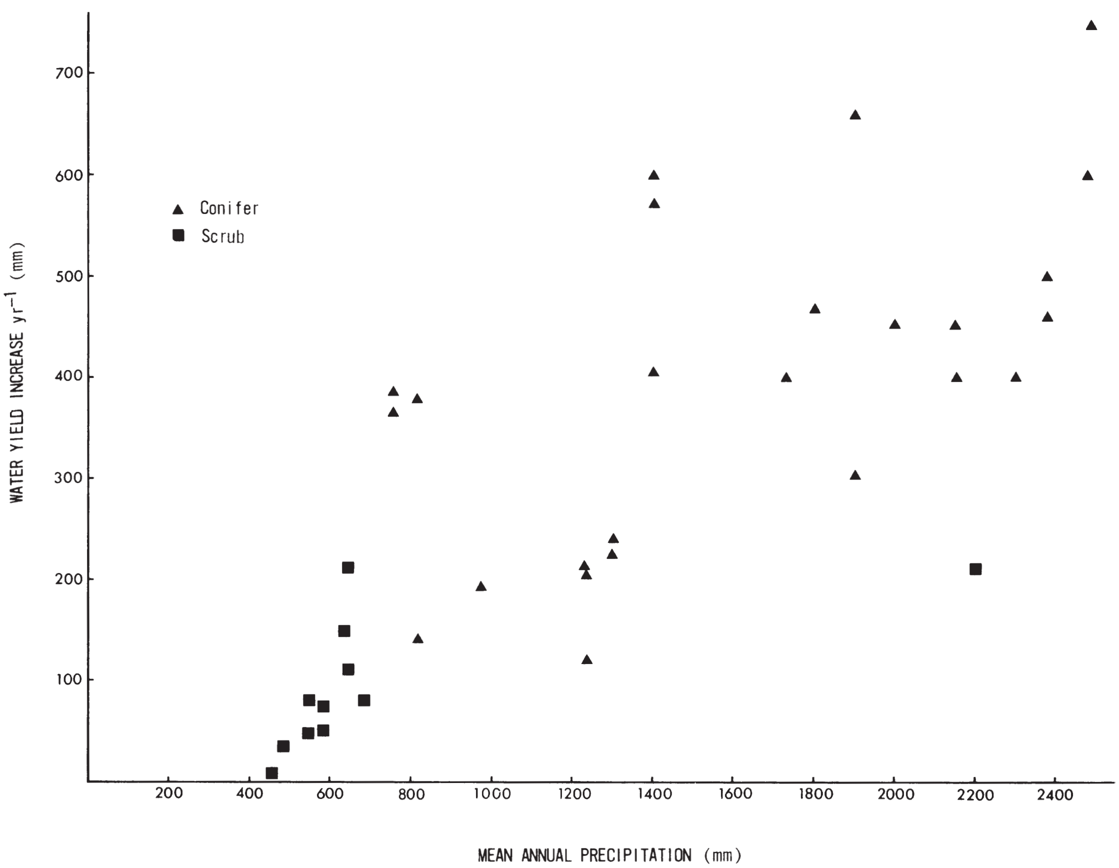


Figure 2.2. First-year increases in water yield after vegetation removal versus mean annual precipitation. Note that 500 mm is approximately 20 inches (from Bosch and Hewlett, 1982).

At Wagon Wheel Gap, for example, the average annual precipitation is only 21 inches, the mean pre-treatment water yield was 6.1 inches, and clearing one of the two catchments increased annual water yields by an average of only one inch. At Fool Creek in the Fraser Experimental Forest the average annual precipitation is approximately 26 inches, the average annual runoff prior to harvest was 8.7 inches, and the average increase in flow from cutting 50% of the vegetation or 40% of the watershed was 3.1 in, or approximately three times the value observed at Wagon Wheel Gap. The water yield increases obtained from other watershed experiments in the FEF and Coon Creek in south-central Wyoming are quite consistent, as they are in similar forest types and exhibit similar hydrologic behavior. There is greater variation in the increases in runoff measured from the paired-watershed studies at Beaver Creek, as there is more variation among these watersheds and they exhibit greater variability in the processes controlling the amount and timing of runoff (Baker, 1986).

In summary, a certain amount of evaporation will occur independent of the vegetative cover. Until annual precipitation exceeds this threshold, a change in vegetation generally will have no effect on annual water yields as long as the basic runoff processes are not changed (e.g., the infiltration rate and soil moisture storage capacity are not altered by compaction, paving, or soil erosion).

As annual precipitation exceeds this threshold, the vegetation plays an increasingly important role in the water balance equation by intercepting rain and snow and transpiring water. An increase or decrease in the density of the vegetation cover – as indexed by basal area or leaf area – will have a corresponding effect on runoff. Both equation 2 and paired-watershed studies in the Rocky Mountains show that the threshold for forest management to affect water yields is approximately 18-19 inches, and this threshold corresponds with the threshold of 18-20 inches suggested by Bosch and Hewlett (1982) (Figure 2.2).

The only way to reliably increase water yields in areas that receive less than 18-19 inches of annual precipitation is to reduce infiltration by paving or other land surface treatments and collecting the resulting runoff.

2.2.2. Timing of an increase in runoff.

The timing of an increase in water yield depends on the timing of soil moisture recharge. In environments with dry summers and wet rainy winters, most of the water yield increase after forest clearing shows up in the fall and early winter because the reduction in summer transpiration causes the soils to be wetter at the beginning of the rainy season (e.g., Harr et al., 1975; Ziemer, 1981). Hence less rain is needed to recharge the soil and groundwater at the beginning of the wet season, and more of the initial rainfall in the winter wet season is converted into runoff. Later in the winter the proportional increases in runoff are much smaller, as once the soil is recharged the presence or absence of trees will primarily affect the amount of rain lost to interception. While annual interception losses in coniferous forests are generally in the range of 20-30% (Dunne and Leopold, 1978), the percentage of rainfall lost to interception during large storm events is much less (Zinke, 1967). Hence an increase or decrease in the density of the forest vegetation has a proportionally smaller effect on runoff once the soils have been fully recharged.

In snow-dominated areas the basic water balance equation still applies, but there are some important differences with respect to the timing of the observed water yield increases. In Colorado nearly all of the water yield increase occurs in early spring, and the difference in the timing of the water yield increase is due primarily to the difference in the timing of soil moisture recharge. As in rain-dominated areas, removal of the forest canopy results in less soil water depletion during the summer, so less water is needed for soil moisture recharge and more of the early snowmelt is converted into runoff. The removal of the forest canopy also increases the rate of spring snowmelt, so the increased snowpack typically does not extend the duration of spring snowmelt. The net result is that in snowmelt-dominated areas the increase in runoff due to forest harvest occurs primarily in spring, on the rising limb of the snowmelt hydrograph, rather than in the fall and early winter. This pattern is clearly shown by comparing the average annual hydrographs prior to and after timber harvest for the snowmelt-dominated Fool Creek watershed on the Fraser Experimental Forest (Figure 2.3). Month-by-month comparisons of the pre- and post-treatment flows for Fool Creek confirm that May is the only month with a consistent, statistically-significant increase in runoff (Troendle and King, 1985). A significant increase in June runoff can occur in places where, or years when, peak snowmelt occurs slightly later in the year (Troendle et al., 2001). As discussed in section 2.2.5, an increase in soil moisture after timber harvest has no detectable effect on low flows in Colorado.

2.2.3. Causes and variability of water yield increases after forest harvest.

In most forested areas the primary effect of forest harvest or afforestation is to alter the amount of growing season

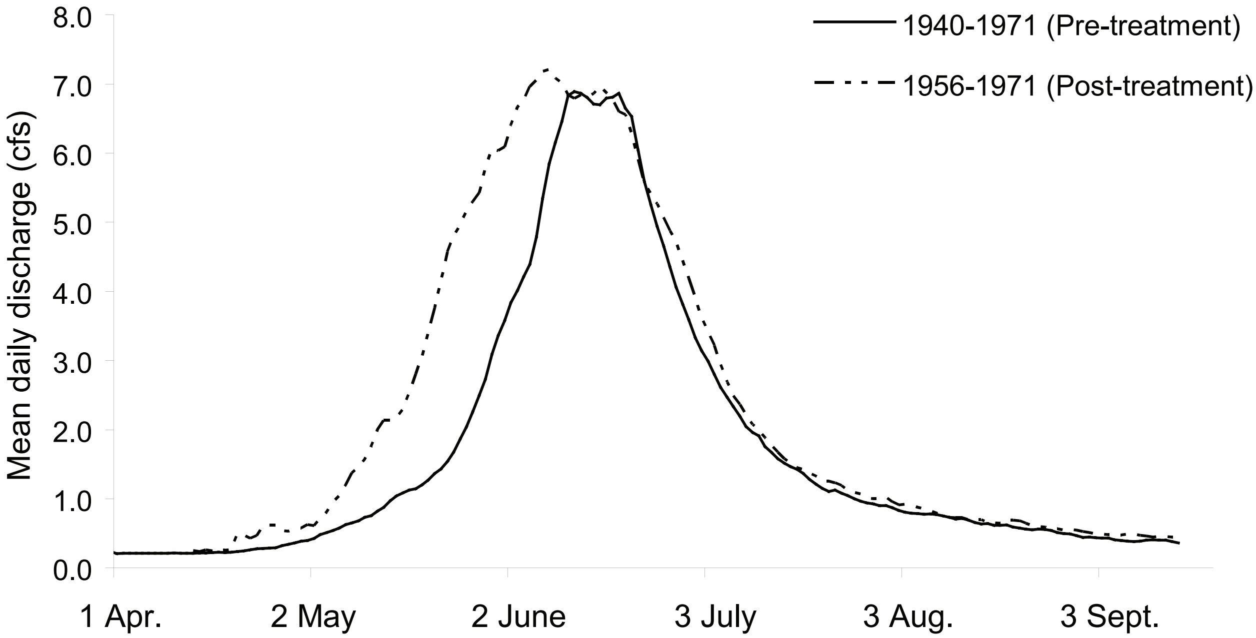


Figure 2.3. Mean annual pre- and post treatment hydrographs for Fool Creek, Fraser Experimental Forest.

ET, and this change in ET drives the change in runoff in accordance with equation 1. However, the snow-dominated areas of Colorado are unusual in that winter interception is a much larger component of the overall water balance than for most other forested areas. Studies on the FEF have shown that completely removing the tree canopy increases the amount of snow water equivalent (SWE) on the ground by approximately 20-45%, depending on aspect. Clearcuts on the Manitou Experimental Forest increased the SWE by 8-35% (Gary, 1975).

The decrease in winter interception due to forest harvest can be as large or larger than the decrease in summer ET, particularly in wetter years. The exceptionally high winter interception rates at the FEF can be attributed to the low relative humidity, high average wind speed, large surface area of snow in the tree canopy, and the nearly continuous presence of snow in the tree canopy in winter, especially on north-facing slopes. A series of detailed studies have shown that the observed increase in SWE following forest harvest or thinning is due almost entirely to the reduction in winter interception rates rather than the redistribution of snow into openings as claimed in some earlier studies (Troendle and King, 1987). For a given vegetation type and aspect the amount of winter interception is nearly a fixed percentage of winter snowfall, and the increase in peak SWE is directly proportional to the amount of the canopy that is removed. In the subalpine zone the mean percent interception loss in winter and spring is approximately 25% for conifers and 11% for aspen (Troendle et al., 2003). In lower elevation areas the increase in SWE after timber harvest is generally smaller, and a 22% reduction in basal area did not significantly increase the SWE at the Manitou Experimental Forest (Gary, 1975).

The other factor that governs the change in water yield is the amount of evaporation and transpiration in the growing season. The general principle is that in drier years and areas with shallower soils, summer ET savings are minimized because the combination of soil evaporation and transpiration from the residual vegetation will use nearly all of the available water. Under these conditions the reduction in winter interception will provide nearly all of the increase in water yield from forest harvest. In wetter years and on aspects with less winter interception, proportionally more of the increase in water yield will be derived from the reduction in summer ET (Troendle and King, 1985; Troendle and Reuss, 1997).

Plot-scale data for different levels of thinning in lodgepole pine indicate that the reduction in winter interception on the FEF is approximately two to three times larger than the reduction in summer soil water depletion (Wilm and Dunford, 1948). For the Fool Creek Experiment it has been estimated that, on average, approximately 50% of the increase in water yield is due to the reduction in snow interception, and 50% is due to lower ET during the growing season (C. Troendle, Matcom Corp., pers. comm., 2000).

Plot- and catchment-scale studies on the FEF indicate that the pattern of forest harvest has a surprisingly small effect on the magnitude of the resulting water yield increase. In the Deadhorse North experiment 36% of the basal area was removed by patch cutting, and the increase in annual water yield was proportional to the percent of basal area removed. Removing 40% of the basal area in the first step of a shelterwood cut resulted in a 16% increase in peak SWE, and the observed increase in streamflow, while too small to be statistically significant, was consistent with the observed increase in SWE (Troendle and King, 1987). These results suggest that the reductions in winter interception were directly transformed to streamflow, and the additional water generated in a spatially-distributed manner across the hillslopes was able to reach the stream channel.

In drier areas and areas where summer ET is the primary source of "saved" water, the pattern and location of forest harvest can have a much greater effect on the observed change in annual water yields. Baker (1986) found that watersheds with longer and flatter hillslopes generated smaller increases in runoff after forest thinning than steeper and narrower watersheds. Other studies have suggested that harvest-induced increases in runoff can be reduced because the remaining trees can scavenge the "saved" water from upslope areas before it is able to reach the stream channel (e.g., Ziemer, 1968; MacDonald, 1987). In general, the pattern of forest harvest and the location of the harvest relative to the stream channel are more likely to affect the magnitude of a water yield increase in drier, lower-elevation areas and possibly in drier years in sub-alpine areas. In the higher-elevation, wetter sub-alpine zone most of the interception savings will probably be transformed into runoff, regardless of the pattern of harvest or proximity to a stream channel.

Aspect can affect the magnitude and timing of potential water yield increases in higher-elevation forests by affecting both the amount of sublimation and the rate of snowmelt. In the sub-alpine zone, north-facing slopes typically have denser vegetation than south-facing slopes. This denser vegetation has higher interception rates, as the greater leaf area and higher winter branch turgor capture and hold more snow. Recent modeling studies assume a winter-spring interception loss rate of 32% for north-facing slopes, 26% for east- and west-facing slopes, and only 17% on south-facing slopes (Troendle et al., 2003). The difference in winter interception rates results in a greater potential to increase water yields on north-facing slopes than south-facing slopes (Troendle et al., 1994). In contrast, aspect has relatively little effect on the magnitude of the reduction in summer ET after forest harvest, as the amount of ET is limited primarily by the amount of water rather than the amount of incoming energy (Troendle et al., 1994).

In the case of Deadhorse Creek, openings on south-facing slopes had a smaller increase in SWE than expected from interception studies on other aspects (Troendle and King, 1987). This suggests that the orientation of forest openings relative to the sun can affect the increase in SWE resulting from forest harvest, and this effect is probably due to differences in incoming shortwave, reflected shortwave, and longwave reradiation from the surrounding leave trees. In general, forest harvest will increase melt rates by increasing the amount of incoming solar radiation, increasing the turbulent transfer of sensible heat to the snowpack surface, and increasing the transfer of latent heat by condensation and freezing. All of these factors will vary with aspect, slope, prevailing wind direction, and wind speed. These differences mean that the increase in SWE, the increase in water yield, and the timing of an increase in water yield will vary with aspect (Baker, 1986).

The size of the openings created by forest harvest and the amount of roughness also can affect the magnitude of the increase in SWE after forest harvest. Larger openings are more subject to wind scour, but this effect can be ameliorated if there is sufficient slash or other sources of roughness to capture and hold the snow within the opening (Troendle and Meiman, 1984). Because the annual runoff in snow-dominated areas is closely related to the maximum SWE, the ability of forest openings to retain snow can directly affect the magnitude of the increase in water yield after forest harvest.

The water balance equation and the interplay between evaporation, interception, and transpiration can explain the observed interannual variability in water yield increases in response to forest harvest. In dry years a greater proportion of the precipitation is needed to recharge soil moisture, and there may also be slightly greater soil moisture depletion due to the increase in potential evapotranspiration. In wet years there will be more soil moisture carryover, and less snowmelt or winter precipitation is needed for soil moisture recharge. Hence more of the interception and transpiration savings can be converted into runoff. The net result is that the increases in runoff following forest harvest are substantially greater in wet years than in dry years (Baker, 1986; Troendle and Kaufmann, 1987).

On Fool Creek the annual water yield increase ranged from 1.6 inches in the very dry year of 1963 to 6.4 inches in the exceptionally wet year of 1957 (Figure 2.4). Figure 2.5 shows that the increase in annual water yields observed at Fool Creek generally is proportional to the estimated runoff in the absence of any forest harvest. Since annual runoff is directly proportional to the annual precipitation, this means that the increase in runoff due to forest harvest in any given year is directly proportional to annual precipitation. In the case of Fool Creek, approximately 40% of the observed increase in annual water yields can be attributed to the variation in the amount of SWE on 1 April (Troendle and King, 1985). The remaining scatter in Figure 1 can be attributed to differences in seasonal precipitation, snowpack sublimation, and numerous other factors that affect the amount of runoff.

The tendency to have larger water yield increases in wetter years was also apparent in the Wagon Wheel Gap study. Here the water yield increase in a wet year was twice as large in an average year, and the water yield increase dropped to zero in years with below normal precipitation (Bates and Henry, 1928). A series of paired-watershed studies on ponderosa pine stands in Arizona also showed that forest harvest caused much larger increases in water yields in wet years than dry years (Baker, 1986).

2.2.4. Rate of hydrologic recovery.

The values in Table 2.1 and the water yield increases discussed to this point all represent the average change that would be expected in the first year after harvest. In the absence of any other management activities, these increases in runoff will decline over time with forest regrowth (Figure 2.6) (Troendle and King, 1985). Colorado's relatively cold, dry climate means that forest regrowth is slow relative to most other areas, and one would therefore expect slower hydrologic recovery rates. The long-term snowpack and streamflow records from the FEF allow a more rigorous quantification of hydrologic recovery in the sub-alpine forest than most other areas (e.g., Troendle and King, 1985). These basin-scale hydrologic data are complemented by more detailed studies on the rate at which leaf area, basal area, and other key characteristics recover following forest harvest (e.g., Kaufmann, 1985).

Since the Wagon Wheel Gap study was only monitored for seven years after harvest (Bates and Henry, 1928), the longest-running paired-catchment study is the Fool Creek experiment. After forty years annual water yields

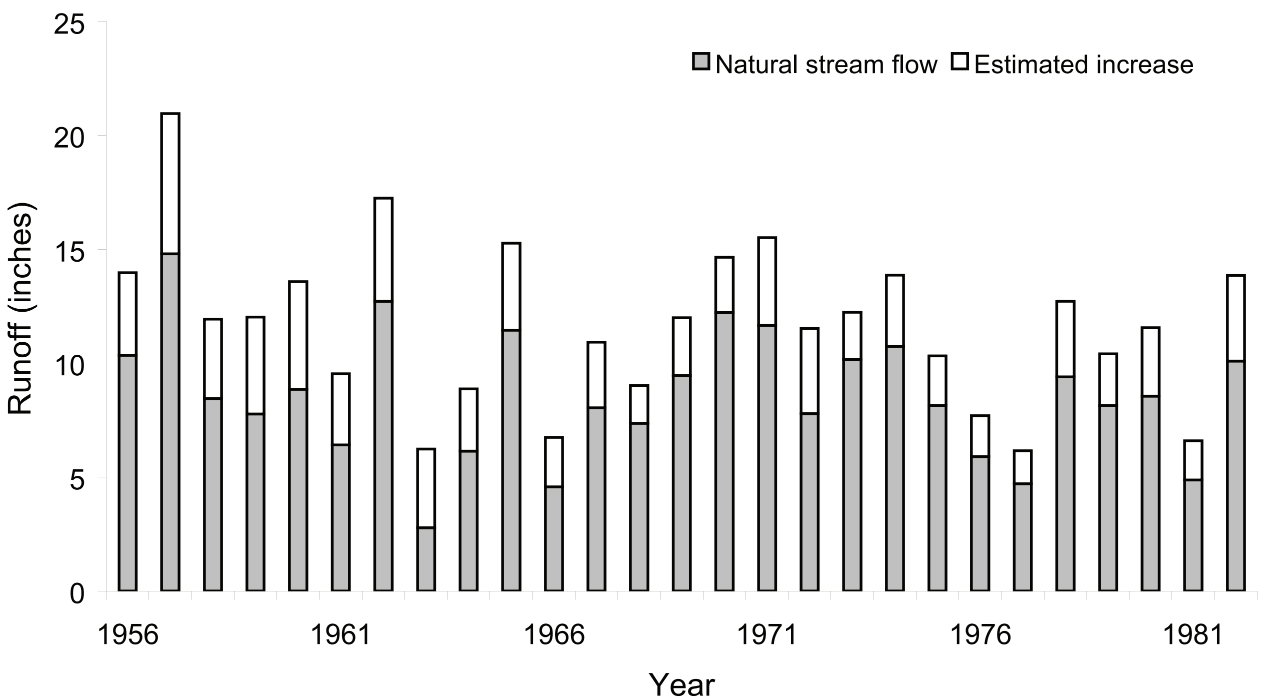


Figure 2.4. Natural and increase in water yields by year for the Fool Creek catchment for the first 27 years after harvest.

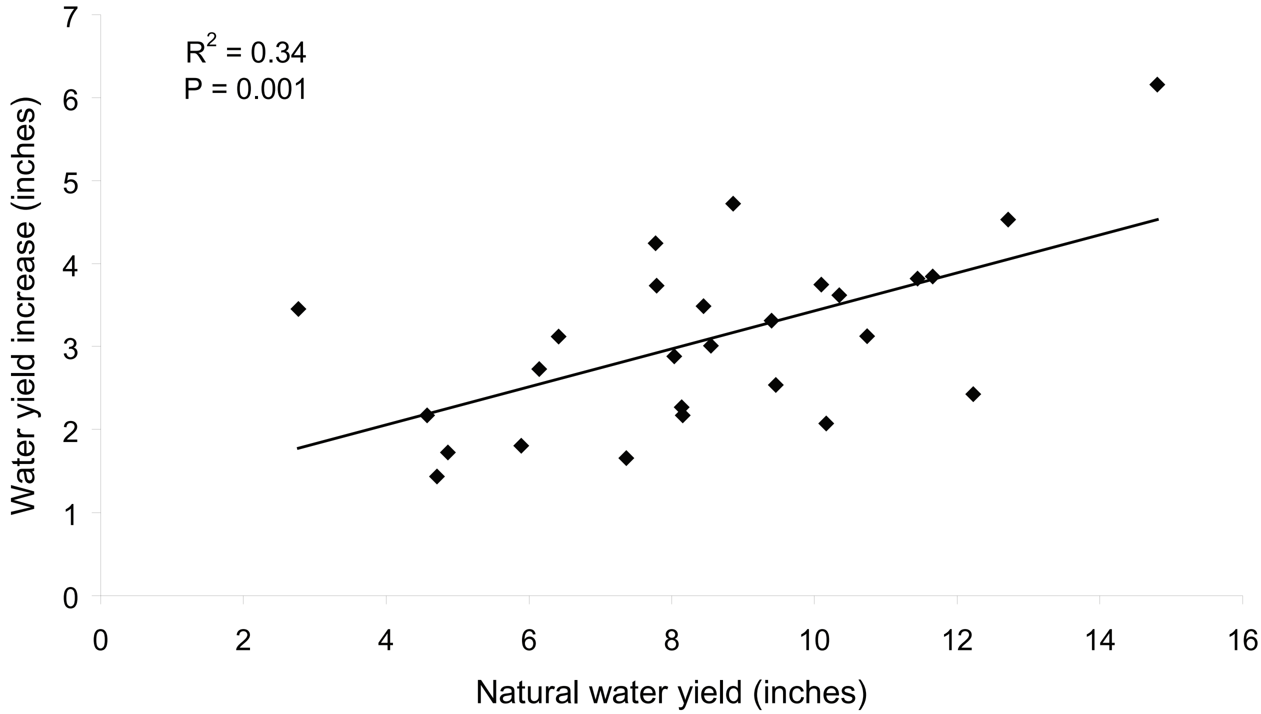


Figure 2.5. Harvest-induced increases in water yield from Fool Creek versus the predicted natural water yield in the absence of any forest harvest. Data are from 1956-1982.

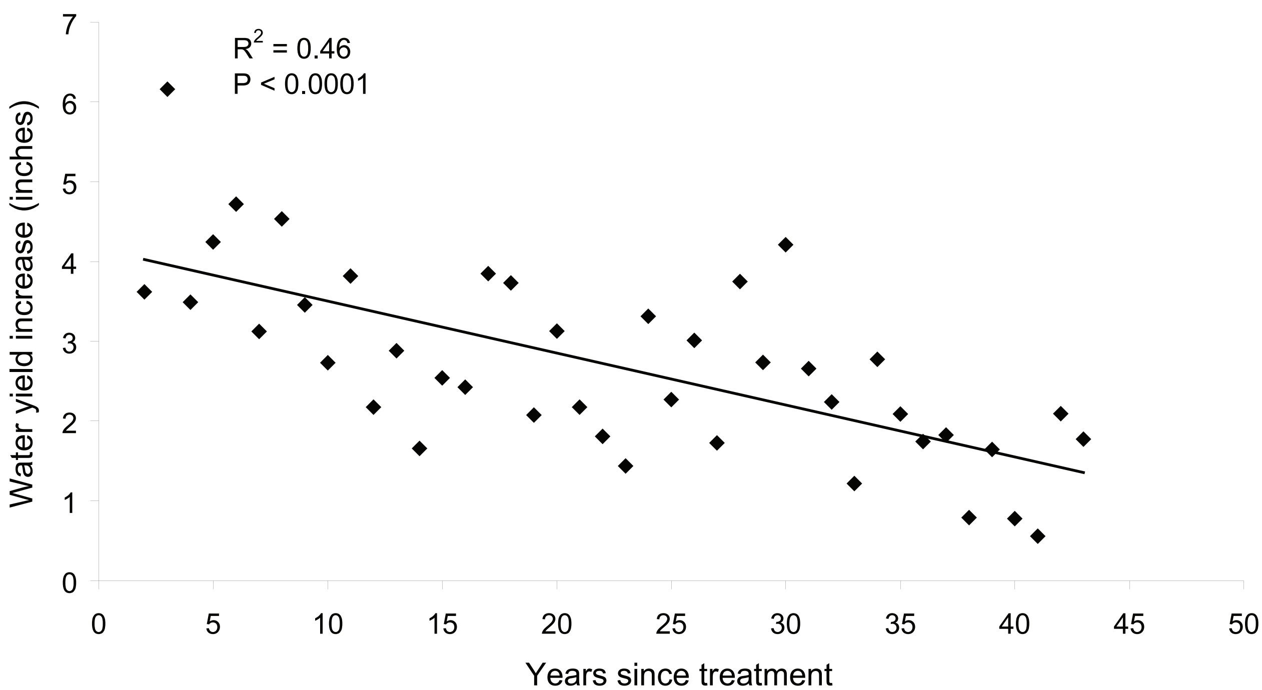


Figure 2.6. Decline in the post-harvest increase in annual water yields over time, Fool Creek, Colorado.

are still elevated relative to the control catchment. Published data show a linear decline in water yields from 1958 to 1986 (Figure 2.6) (Troendle and Nankervis, 2000), and this linear decline has continued through 2000 (C. Troendle, Matcom Corp., pers. comm., 2003). The data suggest annual water yields will return to their pre-treatment values in approximately 60 years.

Hydrologic recovery is expected to be faster in most other vegetation types relative to the subalpine spruce-fir forest type. Hydrologic recovery has been estimated to be on the order of 15-45 years for species that resprout or are faster growing, such as aspen (Troendle and Nankervis, 2000). Hydrologic recovery also should be faster in areas with less annual precipitation, as less regrowth is needed to return summer water losses to pre-harvest levels. Paired-watershed studies in the ponderosa pine zone in Arizona showed that harvest-induced water yields persisted for only seven years on the watershed that was completely clearcut, and from three to seven years for three of the four watersheds that were partially cleared. Approximately 10 years were needed for hydrologic recovery on the watershed that was 77% cleared, and the longer treatment effect on this watershed was attributed to the combination of north-facing aspects and shorter slopes (Baker, 1986). Paired-watershed studies in Oregon also have shown that the time to hydrologic recovery may be reduced if fast-growing riparian species, such as alder, become more prevalent after forest harvest, or there are species that resprout. A similar effect might be expected in Colorado. In some cases the replacement of mature or over-mature trees with younger, more vigorous vegetation can cause water yields to drop below the pre-harvest values after 10-20 years (e.g., Vertessey et al., 1996), and it is not known whether an analogous decline in annual water yields will occur in Colorado.

2.2.5. Effects of vegetation change on the size of peak flows.

The effect of forest management on the size of peak flows is an important concern for resource managers and the public. An increase in the size or duration of high flows can increase the sediment transport capacity, alter channel geometry by scour or bank erosion (Schumm, 1971), and raise water levels (stage) in the affected streams. Hydrologic theory suggests that any reduction in forest cover will have a progressively smaller effect on peak flows with increasing flow magnitude or recurrence interval, and most reviews on the effects of forest management on runoff have come to a similar conclusion (e.g., Anderson et al., 1976; Austin, 1999). The underlying logic is that during the largest rain or snowmelt events the soils and vegetative canopy will have little additional storage capacity, and under these conditions much of the rainfall or snowmelt will be converted to runoff regardless of the amount or type of vegetative cover.

If the amount and delivery of runoff are not significantly altered by roads or soil compaction, the primary hydrologic change in rain-dominated areas is a reduction in interception. Since interception losses should be proportionally less in the larger storms (Zinke, 1967), one would expect proportionally smaller increases in the larger peak flows. An analysis of daily flows from 28 paired-catchment experiments showed that the median increase in the 95-99th percentiles of daily flows (i.e., the flows that are equaled or exceeded for 3-20 days per year) was about 10-15% (Austin, 1999). The effect of forest harvest on larger flows (i.e., >2-year recurrence interval) in rain-dominated areas is much more difficult to discern because of the variability between basins and the small number of events for analysis. The effects of forest management on runoff from these extreme rain events is still a matter of considerable controversy (e.g., Jones and Grant, 1996; Thomas and Megahan, 1998; Jones, 2000).

In the mixed conifer zone in northern Arizona the removal of approximately 28% of the basal area increased the smaller peak flows (97.5th percentile) by 16% and the larger peak flows (99th percentile) by only 6% (Austin, 1999). In northern Arizona the peak flow from a 100-year storm event was estimated to have increased by 20-28% in watersheds where 30-50% of the timber had been removed (Brown et al., 1974). For this same storm, the estimated increase in peak flow was approximately 90% for a watershed where 77% of the timber had been removed and another watershed where the vegetation had been converted from forest to grass. The largest increase (170%) was on the watershed that had been clearcut and subjected to 100% ground disturbance (Brown, 1974).

In snowmelt-dominated areas the effects of forest harvest on peak flows is more consistent, but one must still be careful to define the peak flow of concern and whether the increase is in relative or absolute terms. In general, forest harvest in snowmelt-dominated areas in the Rocky Mountains will increase the size and frequency of the larger flows, but there is little evidence for an increase in the highest instantaneous peak flows (i.e., flows with a recurrence interval greater than two years). The change in the size of peak flows in snowmelt-dominated areas is due to multiple factors, and these include an increase in the amount of shortwave radiation that reaches the snowpack surface, an increase in the turbulent heat transfer as a result of higher wind speeds at the snowpack surface, an increase in latent heat transfer due to condensation and freezing at the surface of the snowpack, and possibly an increased persistence of the snowpack as a result of the increased peak SWE.

Figure 2.2 showed the effect of forest harvest on the average annual hydrograph for the Fool Creek catchment in the FEF. On average, the annual daily maximum peak flow increased by 23% as a result of removing 50% of the forest cover (Troendle and King, 1985). The average number of days with flow at or above bankfull increased from 3.5 to 7 days per year (Troendle and Olsen, 1994). A reanalysis of data from the Wagon Wheel Gap study showed that the annual daily maximum flow increased by 50% (Van Haveren, 1988), and a similar increase in the size of the annual snowmelt peak occurred as a result of the shelterwood cut on the North Fork of Deadhorse Creek (Troendle and King, 1987). These and other studies indicate that complete removal of the forest canopy in small watersheds in the sub-alpine zone will increase the size of the annual daily maximum flow by about 40%. If only part of the forest canopy is removed, the increase in the size of the annual daily maximum flow is directly proportional to the amount of basal area removed.

Increases in the instantaneous annual maximum flow are more variable (e.g., King, 1989). In general, harvest-induced increases in the instantaneous annual maximum flow should be similar to, or slightly smaller than, the change in the annual daily maximum peak flow because forest harvest will have a progressively smaller effect on the highest peak flow (i.e., peak snowmelt is limited by the amount of energy that can be delivered to the snowpack). For example, forest harvest had no detectable effect on the three largest instantaneous peak flows on Fool Creek despite the significant increase in the annual daily maximum peak flow (Troendle and Olsen, 1994).

Harvesting 24% of the 4,100-acre Coon Creek watershed increased both the annual daily maximum and the instantaneous peak flows resulted by about 8%, but these increases were not statistically significant (Troendle et al., 2001). However, there was a significant increase in the 70th to 90th flow quantiles, where the value nearest zero represents the lowest flow on record and the value closest to 100 represents the highest flow on record (Troendle et al., 2001). The absence of a statistically significant increase in the annual peak flows in the Coon Creek study as compared to Fool Creek is partly due to the much smaller number of pre- and post-treatment data points, the variability in the relationship between the treated and the control basins, and the fact that only 24% of the Coon Creek basin was harvested. Peak flow increases also may be smaller in larger basins with a wide range of elevations and aspects, as the peak flows from different parts of the basin may not be synchronized.

The effect of forest harvest on the size of peak flows is much more complicated and controversial in areas where the largest peak flows are caused either by mid-winter rain-on-snow (e.g., Harr, 1986) or rain-on-spring-snowmelt (MacDonald and Hoffman, 1995). Since rain-on-snow events are rarely encountered in Colorado and don’t appear to be a primary cause of the peak flows above 7500 ft (Jarrett, 1993) or the largest runoff events below 7500 ft (N. Doesken, Colorado State University, pers. comm., 2000), the effect of forest management on rain-on-snow events will not be considered further.

Some forest hydrologists have suggested that watershed-scale increases in the size of peak flows can be exacerbated or ameliorated by harvesting different portions of the basin to alter the timing of runoff from different tributaries (e.g., Harr, 1981). The argument is that forest harvest close to the mouth of a basin may accelerate runoff and help desynchronize the runoff from the lower portion of a basin relative to the upper portion. Alternatively, forest harvest in the upper portions might accelerate the delivery of water to downstream areas and further increase peak flows by synchronizing the peak flows from the upper and lower portions of a basin.

Research in Colorado and other snowmelt-dominated areas indicate that forest harvest usually has little or no effect on the timing of the peak flows. Forest harvest caused no significant change in the timing of the snowmelt peak at Wagon Wheel Gap, Deadhorse North, Deadhorse South (Troendle and King, 1987; Troendle, 1987a), or Coon Creek (Troendle et al., 2001). In contrast, the annual maximum peak flow on Fool Creek occurred an average of 7.5 days earlier (Troendle and King, 1985). The earlier peak at Fool Creek is attributed to the faster and earlier melt in the lower parts of the catchment, and this superimposed an earlier and sharper peak on a formerly flat or bimodal snowmelt peak (Figure 2.3). Overall, there is little evidence to suggest that forest harvest can consistently and significantly affect the timing of peak flows in Colorado. Hence the issue of synchronization appears to be of little practical significance, and will not be considered further.

2.2.6. Effects of vegetation change on low flows.

In contrast to changes in the size of peak flows, an increase in the size of low flows is generally considered beneficial, as the additional water may be useful for water supply purposes, increasing instream habitat, or improving water quality (MacDonald et al., 1991). Studies outside of Colorado generally show that forest harvest can substantially increase low flows in percentage terms, but the increase is very small when expressed in absolute terms (Austin, 1999). In most cases the increase in low flows is relatively short-lived, as the increase is dependent on the increase in summer soil moisture and/or groundwater levels, and a relatively small amount of regrowth can eliminate the initial post-harvest increase in soil moisture. Studies in other areas have shown that harvest-induced increases in low flows are eliminated within 5-10 years due to vegetative regrowth, particularly along the stream channels (e.g., Hicks et al., 1991; Vertessey et al., 1996).

For Colorado, data on the effects of forest harvest on low flows are only available for snowmelt-dominated catchments. There is general agreement that most summer rain storms have little effect on summer streamflows, as the amount of rain is usually small relative to the available soil moisture storage. Watershed studies indicate that only about 1-3% of summer precipitation is converted into runoff (Bates and Henry, 1928; Berndt, 1960; Troendle and King, 1985; Troendle et al., 2001), indicating that low flows are primarily a function of soil moisture storage and groundwater.

Studies at the FEF have shown that forest harvest can increase summer soil moisture, and this effect is generally larger in wet years and in areas with deeper soils. However, the observed increases in soil moisture did not significantly increase monthly streamflows on Fool Creek in summer or early fall (Figure 2.3) (Troendle and King, 1985). Troendle and Reuss (1997) found a significant increase in base flows at the foot of a harvested hillslope in West St. Louis Creek in the FEF, but this increase in base flows could not be statistically detected at the catchment scale.

Complete harvest of watershed B at Wagon Wheel Gap increased summer minimum flows by about 10%. In absolute terms this converts to just 0.024 cubic feet per second per square mile (Bates and Henry, 1928). This increase was attributed to the reduction in summer ET and resulting increase in soil moisture on the treated watershed. Austin's (1999) reanalysis of the daily flow data from Wagon Wheel Gap showed summer low flows increased only for the first five years after harvest. However, the persistence of this increase is confounded by the drier conditions after the treated catchment was cleared (Troendle and King, 1985).

An analysis of flow-duration curves from the Coon Creek study indicated a significant increase in flows from the 40th to the 90th percentile, but no detectable change in the smaller flows (i.e., 1st to 40th percentile). A reanalysis of runoff data using flow-duration curves would be a useful and more sensitive test of the changes in summer low flows at Fool Creek and possibly Deadhorse Creek.

The changes in low flows that might occur in other forest types in Colorado can only be estimated in accordance with our understanding of forest hydrology. In general, the increase in runoff from forest harvest decreases with decreasing annual precipitation (Figure 2.2). Some studies have shown a relatively large percentage increase in low flows, but because most of the increase in runoff flow comes during moderate and high flows, the absolute increase in low flows is very small (Hibbert and Gottfried, 1987; Austin, 1997).

Hydrologic theory and studies in other areas also suggest that the persistence of any increase in low flows will be much shorter than the recovery period for annual water yields (e.g., Austin, 1999). The shorter recovery period for low flows is attributed to rapid recovery of summer evapotranspiration rates relative to winter interception rates. As observed for annual water yields, a harvest-induced increase in low flows will be smaller or non-existent in dry years, and the recovery of low flows will be faster in drier areas and areas with shallow soils. It may be of some scientific interest to further explore the effects of forest management on low flows over time and on a larger scale through hydrologic models, but present knowledge suggests that water managers should not expect significant increases in low flows as a result of forest management.

2.3. Effects of Roads on Runoff

The previous section focused on the effects of forest harvest on runoff, and this implicitly presumed that the vegetation was cut without any other disturbance. In reality, forest management activities usually require the construction and use of roads, skid trails, and landings. Many forest roads also provide access for recreation and homeowners. The effects of roads, trails, and associated features must be considered when evaluating the effects of forest management on runoff, although the changes in erosion and downstream sedimentation are generally of even greater concern (Chapter 3). The problem is that most catchment-scale studies combine the effects of roads and forest management, or study the effects of roads for only one or two years prior to forest harvest, and this makes it difficult to clearly separate the effects of roads on runoff from the effects of forest management, particularly at the catchment scale.

The hydrologic significance of roads is due to their potential to alter both the processes by which runoff is generated and the rate at which water is delivered to stream channels. The change in runoff processes is due to the fact that forested areas typically have high infiltration rates and most or all of the precipitation and snowmelt infiltrates into the soil; overland flow is rarely observed (Hewlett, 1982; Troendle, 1987b). In contrast, the infiltration rates for unpaved roads are usually no more than 1 mm or 0.04 inches per hour (e.g., Luce and Cundy, 1994). The low infiltration rate means that most of the rainfall and snowmelt on road surfaces will rapidly run off as infiltration-excess overland flow. If the roads are insloped, the road surface runoff is directed into an inside ditch, and these ditches commonly drain directly into swales or stream channels. This means that roads not only generate more runoff as a result of their low infiltration rates, but also provide a pathway to rapidly deliver this runoff to the stream network (Montgomery, 1994; Wemple et al., 1996).

Roads that are cut into the hillslope also can affect the amount and timing of runoff by intercepting the water that normally flows downslope through the soil mantle (e.g., Megahan, 1972). This intercepted subsurface stormflow is often collected by inside ditches and delivered to the stream network in the same way as the runoff from the road surface. This interception of subsurface flow is important because it transforms slower-moving subsurface flow to faster-moving surface runoff, and rapidly delivers this intercepted subsurface stormflow to the stream network (La Marche and Lettenmaier, 2001). The additional runoff generated from roads can initiate channels where they normally would not be present (e.g., Montgomery, 1994), and the road ditches also can be directly linked to the new and pre-existing channels. The resulting increase in the drainage density (defined as the total length of channels per unit area) increases the rate at which water is delivered to the stream network and this can substantially increase the size of peak flows (La Marche and Lettenmaier, 2001). While these changes in runoff generation and routing have been documented at the site scale and several studies have concluded that roads can increase the size of peak flows at the watershed scale (e.g., Jones and Grant, 1996; Jones, 2000; La Marche and Lettenmaier, 2001), the effect of roads on the size of peak flows is still controversial and largely undocumented at the watershed scale (e.g., Thomas and Megahan, 1998).

Most of the research on road runoff has been conducted in rain-dominated areas, but roads should have similar effects in snowmelt-dominated areas. However, there are several reasons why the hydrologic effects of roads in snowmelt-dominated areas may be less than in rain-dominated areas. First, snowmelt rates are much less than peak rainfall rates, so proportionally more of the snowmelt on unpaved road surfaces will infiltrate. Second, the lag between peak snowmelt and peak runoff for small to moderate-sized basins is typically around 5-12 hours, so the more rapid delivery of water from roads to the stream network will not necessarily coincide with peak snowmelt runoff from the rest of the basin.

Data from paired-basin experiments in snowmelt-dominated areas have failed to demonstrate a change in the size of peak flows due to roads. In both the Fool Creek and Coon Creek experiments the roads were built a couple of years before the timber was harvested, and in each case there was no detectable change in runoff. Since the road network occupied less than 2% of the total watershed in each case, it should not be surprising that road construction caused no detectable change in runoff in the short monitoring period prior to forest harvest (Troendle et al., 2001).

In addition to roads, surface runoff can be generated from roofs, hiking or off-road vehicle trails, and compacted areas such as skid trails and landings. Runoff from these areas is more likely to be routed onto forested slopes rather than into ditches, in which case it has a greater likelihood of infiltrating into the soil. To the extent that this runoff can infiltrate into the soil and is not routed to the stream, one generally would expect these other sources of overland flow to have less of an effect on watershed-scale runoff rates than forest roads.

In summary, unpaved roads, compacted areas, and impervious surfaces can generate overland flow. Road cuts will intercept subsurface stormflow, and the road drainage network may rapidly route this runoff into the stream network. The impact of these changes on runoff at the watershed scale depends on the proportion of the watershed that is affected (Harr, 1986) and the drainage design of the compacted areas. Insloped roads with ditches are usually of greatest concern because they typically collect the runoff and deliver it directly to the stream network. The hydrologic changes due to roads have been documented at the road segment and hillslope scale, but have not been confirmed at the catchment scale. Roads and skid trails have an even greater effect on sediment production than runoff, and this issue is discussed in Chapter 3.

2.4. Effect of Fires on Runoff

The effect of wild and prescribed fires on runoff is highly variable because they can affect both the vegetation canopy as well as the physical properties of the soil. At one level, the effect of fires on runoff is similar to forest harvest, as any reduction in the forest canopy will alter the amount of interception and transpiration. Fires may also consume some or all of the litter and duff layers, and this also can increase runoff by reducing the total water storage capacity at a given site (Kittredge, 1948).

The greater concern is the potential for high-severity fires to alter the surface organic layer and mineral horizons in ways that reduce infiltration and increase runoff and erosion rates. The reduction in infiltration can occur by several processes. High-severity fires consume all of the surface litter and duff. Sustained soil heating can burn off much of the organic matter in the top few centimeters of the mineral soil. In this situation there is little protection of the soil surface from rainsplash, and the disaggregated soil particles and ash can clog up the larger soil pores that are crucial for maintaining the high infiltration rates typical of forested areas (Terry and Shakesby, 1993).

The second important change is the generation of a water repellent (hydrophobic) layer at or slightly below the soil surface. Fires can generate a water repellent layer by volatilizing hydrophobic compounds in the organic material, and some of these are driven downwards where they condense on cooler soil particles at or below the soil surface (DeBano, 1981; Letey, 2001). The amount and type of these compounds varies with the type of vegetation, while the depth at which these compounds condense is a function of soil heating. In hotter, slower-moving fires there is more soil heating and these volatile compounds may be deposited at depths of 2-6 inches below the soil surface. In low-temperature and faster-moving fires there is less soil heating and these compounds condense closer to the soil surface. Stronger water repellent layers are associated with increasing burn severity, as high-severity fires vaporize more organic compounds and thereby generate a stronger and more continuous water repellent layer (Tiedemann et al., 1979; DeBano, 1981). The development of a water repellent layer is of concern because this can greatly reduce infiltration rates. Once rainfall saturates the thin layer of ash or soil above the water repellent layer, any additional rainfall will run off as overland flow.

The development of a post-fire water repellent layer has been most extensively studied in chaparral ecosystems, but post-fire water repellency has also been documented under a number of different conifer species, including ponderosa pine (Helvey, 1980), lodgepole pine (Meeuwig, 1971), and Douglas-fir (Helvey, 1980). Post-fire water repellent layers are believed to be less likely in vegetation types, such as aspen, that have less surface fuels and fewer secondary compounds (MacDonald et al., 2000).

Post-fire water repellency is generally believed to be more severe in areas with coarse-textured soils. Coarse-textured soils have a much lower particle surface area than fine-textured soils, and this effectively results in a greater concentration of the water repellent compounds per unit surface area (Meeuwig, 1971; DeBano, 1981).

Changes in soil moisture will also affect the strength of a water repellent layer. If moisture is present, a water repellent soil will slowly wet up due to the strong hydraulic gradient and movement of water vapor (DeBano, 1981). As a water repellent soil wets up, there usually is usually a threshold at which a soil ceases to be water repellent (Crockford et al., 1991; Dekker and Ritsema, 1994; Doerr and Thomas, 2000). Upon drying the water repellent conditions can be re-established (Shakesby et al., 1993).

The persistence of a post-fire hydrophobic layer will depend on the initial strength and thickness of the hydrophobic layer and the animal activity, plant regrowth, and physical and chemical processes that collectively act to break down the hydrophobic layer (DeBano, 1981). Soil water repellency usually returns to pre-burn conditions in no more than six years (Dyrness, 1976; DeBano, 1981), and several studies have documented a much more rapid recovery (e.g., DeByle, 1973; Reeder and Jurgensen, 1979).

Until recently there has been relatively little work on the development and persistence of post-fire soil water repellency in Colorado. In most cases the presence of a water repellent layer has been inferred from the observed post-fire increases in runoff and erosion (e.g., Morris and Moses, 1987). However, in the summer of 2000 detailed measurements of soil water repellency were made on five fires in the Colorado Front Range that burned from 1 to 22 months earlier (Huffman et al., 2001). Strong water repellency was found in ponderosa and lodgepole pine forests that burned at high or moderate severity, regardless of whether the fire was a wildfire or a prescribed fire. Areas that burned at low severity generally had little or no more water repellency than unburned areas. Soil water repellency was strongest at the soil surface, but significant water repellency was found to a depth of just over two inches. The strength of the water repellent layer generally increased with increasing percent sand, and water repellency generally was reduced when gravimetric soil moisture exceeded 15-25% (Huffman et al., 2001; MacDonald and Huffman, in review). Repeated measurements on the Bobcat Fire southwest of Fort Collins have shown a progressive weakening of soil water repellency 3 and 12 months after burning, and this is consistent with the weak water repellency found in the Crosier Mountain fire 22 months after burning (Huffman et al., 2001; MacDonald and Huffman, in preparation). The tremendous spatial variability in soil water repellency after fires makes it difficult to accurately determine the magnitude and distribution of soil water repellency, or to predict the likely effects on runoff at the watershed scale.

This work and other studies indicate that summer convective storms in the first two, or possibly three, years after burning pose the greatest risk for post-fire flooding and erosion (Moody and Martin, 2001; MacDonald et al., 2001b). The combination of high precipitation intensities, high percent bare soil, and a strong water repellent layer can increase runoff rates from severely-burned areas by one or two orders of magnitude (i.e., 10-100 times) relative to unburned areas. Such increases in summer storm runoff have been documented for the 1994 South Canyon fire on Storm King Mountain in western Colorado (Cannon et al., 1998), the 1996 Buffalo Creek fire southwest of Denver (Jarrett and Browning, draft manuscript), and the Bobcat fire southwest of Fort Collins (Lange, 2001; Kunze, 2003).

Other wildfires in Colorado and the Rocky Mountains have not resulted in as large a change in runoff and erosion rates. For example, the 1994 Hourglass fire near Pingree Park did not appear to have the same effect on runoff as the Bobcat or Buffalo Creek fires. Earlier studies in the same area noted little or no changes in runoff after the Comanche wildfire (Delp, 1968; Meyers, 1968). Post-fire runoff from a basin burned in the 1988 Yellowstone fire was consistent with the change expected from forest harvest (Troendle and Bevenger, 1996) rather than the sharp increase in peak flows that were observed after the South Canyon, Buffalo Creek, and Bobcat fires.

The absence of detailed data from most of these sites makes it difficult to determine why there have been such differences in response, but one possible explanation is the relative likelihood of high-intensity rain storms. At lower elevations in the Front Range most of the recent large fires have been followed by relatively extreme precipitation events, while comparable events apparently did not occur after the Hourglass or Comanche fires. Some atmospheric scientists have suggested that large burned areas affect local atmospheric conditions and thereby facilitate the development of high-intensity convective storms. At higher elevations there is a lower likelihood of high-intensity rainstorms. The possible increase in the frequency and magnitude of convective rainstorms after large fires is a topic that should be investigated further, as the empirical evidence indicates that relatively large storm events have occurred after each of the recent large fires in the Front Range except for the first year after Hayman fire, when there was an unusually severe drought.

It is important to recognize that because forest soils have high infiltration rates, a post-fire water repellent layer does not have to be completely eliminated before it becomes hydrologically ineffective. Prescribed fires may have patches burned at moderate or high severity that are strongly water repellent, but these patches are usually limited in size and spatially discontinuous (Huffman et al., 2001). Much or all of the runoff from these water repellent areas is likely to infiltrate further downslope, so the increase in runoff at the watershed scale should be much less for prescribed fires than large wildfires.

In summary, high-severity fires in forested areas can greatly increase runoff rates. The effect of low-severity fires on runoff generally is consistent with the effects of forest harvest on runoff, as low-severity fires do not substantially alter the runoff processes and pathways. Detailed, process-based studies need to be implemented immediately after severe fires in order to better document the magnitude and causes of changes in runoff, and better predict the risk to downstream areas. The observed increase in peak flows after the South Canyon, Buffalo Creek, and Bobcat fires confirm that high-severity wildfires can greatly alter the basic rainfall-runoff response, and there is a need to determine whether similar changes can be expected in other areas of Colorado.

2.5. Threshold of Response, Spatial Scaling, and the Detectability of a Change in Flows

Most of the information in the previous sections was derived from plot or small watershed studies. From a management perspective, however, the question is whether these results can be extrapolated to larger-scale basins. There are effectively four components to this issue, and these are: (1) How much of a forested basin has to be treated in order to generate a detectable change in runoff? (2) Will a given change in runoff be translated downstream? (3) What change in flow is detectable? and (4) Can changes in forest cover cause changes in flows on larger basins, even if the change in flow is not statistically detectable? The answers to these questions have important implications with respect to the larger-scale changes in runoff that are of most interest to resource managers and the public.

The answer to the first question – how much of a basin must be treated in order to detect a change in runoff – is relatively well known. A 1982 review of paired catchment experiments concluded that at least 15-20% of a forested basin must be treated within a short time period in order to detect a change in runoff (Bosch and Hewlett, 1982). Troendle and Leaf (1980) noted that 20-30% of a watershed must be treated to detect a statistically significant change in flow, and this value has been supported by the magnitude of flow changes observed in paired watershed experiments in the Rocky Mountains (Table 2.1). At Coon Creek in south-central Wyoming there was a detectable change in annual water yield as a result of harvest and road-building on 24% of the watershed (Troendle et al., 2001). Small increases in annual water yields were observed by removing 31-33% of the trees in two small watersheds in northern Arizona (Baker, 1986).

Harvesting 36% of the North Fork of Deadhorse Creek caused a significant change in runoff at the sub-watershed scale, but a change in water yields was not detectable at the main weir because the treatment only removed about 5% of the forest canopy above this weir. For ponderosa pine forests in the Black Hills of South Dakota, 25-30% of the forest canopy may have to be removed in order to obtain a detectable increase in annual water yields. This higher threshold is probably due to the drier conditions and relative absence of a strong snowmelt peak.

The second question is whether a flow increase at an upstream location will be delivered to a downstream reservoir, diversion intake, or other location of interest. The general principle is that streams are relatively efficient in conveying water downstream, and this is particularly true in headwater forested areas (e.g., Troendle, 1982). However, as streams flow from the mountains and onto the plains streams may become losing rather than gaining. This means that groundwater is no longer flowing to the streams ("gaining"), but streamflow is seeping out of the channel to support the local groundwater table ("losing"). Hence some of an increase in streamflow due to forest management may be transferred to the adjacent groundwater, particularly in larger basins. The availability of the resultant increase in groundwater for human use will vary from basin to basin. Riparian vegetation and surface evaporation also can reduce the amount of streamflow that is delivered downstream.

The proportion of a change in streamflow that is transmitted to a specific location can vary seasonally and from year to year. During spring snowmelt and early summer streams are more likely to be gaining, while later in the summer groundwater levels tend to drop and streams may shift from gaining to losing (e.g., Graf, 1997). Losses to riparian vegetation and surface evaporation generally will be much less in spring than later in the summer. Thus an increase in spring snowmelt generally should be more readily translated downstream than an increase in low flows. Some of the increases in peak flows may be lost to bank storage and overbank flows, and this will attenuate high flows in the downstream direction. The extent to which an increase in flow is translated downstream will depend on the conditions within that basin, but generally one would expect greater losses with increasing spatial scale and increasing distance from the mountain front.

The third issue is the detectability of a given change. The ability to detect change will vary according to the accuracy of the discharge measurements, the magnitude of the imposed change, the variability of the data, the length of the record prior to and after treatment, the certainty with which we want to detect change (i.e., level of significance), and the certainty of detecting change when in fact there is a change (i.e., power) (Loftis et al., 2001). For small research catchments discharge is typically measured at carefully-designed weirs with impermeable cutoff walls, and in these situations the uncertainty in discharge is generally believed to be on the order of only 2-3%. In contrast, the uncertainty of a single flow measurement in a natural channel is generally believed to be around 5% for moderate flows (Kennedy, 1983) and substantially larger at higher flows (Dickinson, 1967). The extrapolation from individual flow measurements to seasonal or annual water yields may involve a further degradation in accuracy because of the greater error in measuring the highest flows, the uncertainty in the relationship between stage and discharge, and the greater potential for missing data (Kennedy, 1983). Hence the uncertainty of flow measurements at a typical stream gauging station is probably closer to 10%, or several times greater than in a research setting, and this greater uncertainty limits our ability to detect a change in runoff due to a given change in vegetation or land use.

The use of paired-watershed design greatly increases our ability to detect change (Wicht, 1967; Loftis et al., 2001), as the data from a control watershed can be used to remove much of the temporal variability in runoff. The problem is that this design is very difficult to apply in larger watersheds, as it is nearly impossible to find two large basins where one basin can serve as an untreated control and the other basin can be subjected to treatment. An extensive search across several states was needed to find a 2,200-acre control basin and an adjacent unharvested basin that could be used to test the effects of forest management on runoff at an operational scale (Troendle and Nankervis, 2000). The resulting Coon Creek study did show that the hydrologic effects of forest management on a 6.5 mi^2 basin were directly comparable to the results from much smaller basins (Troendle et al., 2001). Results from even larger basins are scarce and not as definitive, as one has to rely on natural disturbances in an uncontrolled experimental design. Love (1955) claimed to detect a two-inch increase in streamflow as a result of extensive beetle kill in the 762 mi^2 White River basin, but his results were contested by Bue et al. (1955). A later study used a different, somewhat unconventional procedure to detect the changes in flow caused by beetle kill in both the White River and the Yampa River, and this claimed that water yields increased by about 10% relative to the Elk River basin (Bethlahmy, 1974). The increases claimed by Love (1955) and Bethlahmy (1974) are consistent with the results obtained on much smaller basins, and support the view that the changes in runoff measured on small experimental basins can be scaled up to much larger basins.

The other factor that greatly affects the detectability of change is the closeness of the relationship between the control and treated basins, or the consistency of the data before and after treatment if observations are limited to a single basin (Loftis et al., 2001). Typically there is a relatively high correlation between paired basins for annual water yields, and a strong correlation was essential to detecting a change in annual water yields at Coon Creek because there were only five years of calibration data and five years of post-treatment data (Troendle et al., 2001).

The correlation in the size of peak flows between paired basins is usually not as strong, and this makes it more difficult to detect the effect of a given treatment on the size of peak flows. In the case of the Coon Creek experiment, it was not possible to detect a change in the size of the average annual maximum peak flow, even though the snowpack data and results from other studies indicated that there probably was an increase (Section 2.2.5). The detection of a significant change over time at a single stream gauging station is much more difficult because there is almost always more unexplained variability relative to a paired-watershed design (Loftis et al., 2001).

Taken together, these issues mean that one generally should not expect to measure significant changes in runoff in larger watersheds as a result of typical forest management activities. Field research, hydrologic theory, and modeling studies all indicate that a change in forest cover will affect streamflows, particularly at higher elevations where there is more precipitation and hence a greater potential to alter interception and transpiration. The resulting changes in runoff will be transmitted downstream, but with increasing drainage area the changes from a given set of activities will become proportionally smaller. The opportunity to generate a detectable change in runoff in larger basins also may be limited by the amount of area that can be treated within a relatively short time period and the intensity of the applied treatments (e.g., thinning as opposed to clearcutting) (Hibbert, 1979; Ponce and Meiman, 1983). High-severity fires are the primary exception to these general principles, as both the change in runoff and the size of the affected area can be relatively large.

These limitations in detecting change do not mean that changes in runoff won't occur in larger basins, or that there is no opportunity to alter runoff through forest management. Changes in runoff will occur with changes in the forest canopy, and a small percentage change imposed on a large area can equate to a large amount of water in absolute terms. The difficulty is that we cannot expect to measure these changes, so the effect of changes in forest management on runoff at lager scales will usually have to be quantified through hydrologic models, and presumed to be present even though the changes may not be statistically detectable at existing gauging stations.

In conclusion, this chapter has summarized the type and magnitude of changes in runoff that can occur as a result of management actions and wildfires in forested areas. The process-based understanding provided in this chapter is necessary for predicting future changes, as the hydrologic effect of a given action or disturbance depends on a large number of site-specific factors and will vary with changing climatic and site conditions. The next chapter reviews the effects of forest management, or the absence of forest management, on water quality. This is followed by an assessment of the historic changes in forest vegetation in Colorado and the estimated impact of the changes in vegetation on runoff.

Chapter 3
Effects of Forest Management Activities and Fire on Water Quality

3.1. Introduction

Water from forested watersheds is typically higher in quality than waters draining from any other major land use (USEPA, 1980). The natural nutrient cycling and streamflow generation processes in forested areas typically result in water that is low in nutrient and suspended sediment concentrations. Forested watersheds are often the source of public drinking water supplies, and this is particularly true in Colorado due to the higher water yields from forested areas (Chapter 2) as well as the high quality of the runoff.

The potential effects of forest management activities on water quality are often of greater concern than the changes in runoff. Numerous studies have shown that forest management activities may alter the physical and chemical characteristics of runoff. Parameters of particular concern include turbidity, total suspended solids, temperature, and nutrients. Forest management also can affect the amount and transport of larger-sized particles (bedload). Changes in these parameters can affect the beneficial uses of water and aquatic ecosystems (MacDonald et al., 1991). A change in the chemical and physical characteristics of water, especially when the water is being used for domestic purposes, may necessitate changes in water treatment and purification. Increases in other parameters, such as temperature or bedload, may be of less concern for drinking water providers, but may be of greater concern for fisheries or other aquatic resources. Large increases in sediment loads will accelerate the rate of reservoir sedimentation and can initiate a series of changes in channel morphology through channel widening, or increase the likelihood of flooding, which in turn can adversely affect people and aquatic resources.

There is an extensive literature on the effects of forest management on water quality, but only limited information from Colorado. A review of the literature indicates considerable variability with respect to the effects of forest management on water quality. This can be attributed to differences in the type and location of management activities as well as site-specific differences in runoff and erosion processes. This chapter will review the potential effects of forest management activities and fire on erosion and sedimentation, stream temperature, and nutrients. It should be recognized that other activities in forested watersheds, such as urbanization, water diversions, gravel mining, and hardrock mining, may have a greater effect on water quality and aquatic resources than forest management. Because these other activities represent a change in land use rather than forest management, they fall outside the scope of this report and will not be discussed here. The effects on water quality of other types of forest disturbance, such as insects and disease, are not well documented, but the likely effects can be extrapolated from the principles and data presented in this chapter.

3.2. Erosion and Sedimentation

Soil erosion is defined as the detachment and movement of soil particles. The site characteristics that have the greatest effect on erosion rates include the amount and type of precipitation, vegetative cover, soil texture, and slope (Renfro, 1975; Falletti, 1977; Renard et al., 1997). Undisturbed forested watersheds typically have very low erosion rates because of the high infiltration rates and limited surface runoff (Binkley and Brown, 1993a, 1993b). Erosion rates have been estimated as less than 0.1 tons per acre per year for most forested areas in the interior western U.S. (Patric et al., 1984). Similar or lower values have been documented for different forest types in Colorado (Leaf, 1970; Gary, 1975) and for ponderosa pine-mixed conifer forests in northern Arizona (Brown et al., 1974). Average long-term erosion rates can be substantially higher where gullying, debris flows, or other types of mass movements are important erosion processes (Gary, 1975; Bovis, 1978). Undisturbed forested areas in Colorado typically have very low erosion rates because much of the precipitation falls as snow, infiltration rates are high, and mass movements are either infrequent or relatively inactive. In the lower- and mid-elevation forests extreme rainstorms can generate much higher sediment yields. Sediment yields of up to 6 tons per acre were recorded from a small, minimally-disturbed ponderosa pine watershed in northern Arizona that was subjected to a 100-year rainstorm (Brown et al., 1974).

Site or soil disturbance by forest management activities will generally increase soil erosion. Greater site disturbance is usually associated with a greater increase in erosion rates. The type and magnitude of erosion will depend on the amount of soil exposed by management practices, the effect of management activities on infiltration rates, slope steepness, soil type and thickness, amount and intensity of precipitation, and the type of mitigation treatments applied after the disturbance (e.g., seeding, mulching, or ripping) (Swank et al., 1989). Studies in Colorado and elsewhere suggest that erosion rates are acceptably low when there is less than about 30% bare soil (Gary, 1975; Benavides-Solorio, 2003).

With respect to forest harvest, the site disturbance from tree felling is generally considered to be minor. The activities of greater concern are the transport of the logs to a central site, site preparation activities after harvest, and the network of roads, skid trails, and landings used to access the timber and remove it for processing.

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. The amount of site disturbance and compaction, and hence the amount of surface erosion, 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 the use of wheeled skidders can result in more compaction than crawler tractors (Bell et al., 1974; Davis, 1976). In the case of the Deadhorse Creek study, wheeled skidders caused more disturbance in steeper areas because they had to drop their blade to control their downhill speed (C. Troendle, pers. comm., 2001). Cable logging systems result in less site disturbance because yarding trails are only established to the yarding tower machinery and the tower is placed on roads. Cable systems can be ranked in order of decreasing soil disturbance as follows: single drum jammer, high lead cable, skyline, and balloon (Stone, 1973; Brown et al., 1976; Davis, 1976). Helicopters and balloons minimize site disturbance, but are more costly.

Some form of site preparation may be needed to ensure adequate regeneration after timber harvest. The purpose of site preparation is to provide optimal conditions for seed or seedling survival and growth. The two main goals of site preparation are to provide a mineral seedbed and control less desirable, competing vegetation. Site preparation treatments include fire, herbicide application, slash windrow, roller chopping, or other mechanical techniques. Fertilizers can be applied to improve seedling establishment and growth, but fertilization in forested areas is not a common practice in Colorado.

Long-term erosion rates for forest management are typically much lower than other land uses, such as agriculture, because the duration, frequency, and amount of disturbance are much less. Typical timber harvest activities may only increase erosion rates by 0.05 to 0.25 tons ac^{-1} yr^{-1}. More intensive site preparation treatments such as slash windrowing, stump shearing, or roller chopping may increase soil erosion rates by several orders of magnitude to around 5 tons ac^{-1} yr^{-1}. Once a site is revegetated erosion rates rapidly decline to pre-treatment levels (USFS, 1981; Stednick, 2000).

Data from a series of ponderosa pine watersheds in northern Arizona showed that annual post-harvest sediment yields ranged from 0.11 to 1.3 tons per acre in a watershed that was 31% harvested, and only 0.03-0.3 tons per acre for a watershed that was 77% harvested. A third watershed that was clearcut and had 100% of the ground disturbed produced from 0.01 to 27 tons per acre per year (Brown et al., 1974).

A variety of best management practices (BMPs) are commonly applied to minimize the adverse effects of timber harvest on runoff and erosion. The amount of soil disturbance due to yarding by tractors or wheeled skidders can be greatly reduced by operator training and the careful layout of skid trails (Rothwell, 1971). Post-harvest runoff and erosion from skid trails can be greatly reduced by installing water bars, or ripping and seeding compacted areas.

Riparian management zones or vegetative buffer strips are commonly applied along streams and around wetlands and lakes to minimize the potential adverse effects of forest management on aquatic resources. A major purpose of these buffer strips is to minimize the delivery of the eroded soil to the channel network, as in most cases only a small fraction of the total erosion within a watershed reaches streams and other aquatic ecosystems (Walling, 1983). As long as the flow is not concentrated into channels, the vegetation and surface roughness within the filter strip should reduce the velocity of any overland flow (Campbell, 1984). The decreased velocity allows sediment to settle out. In most cases the undisturbed soils in the buffer strip also will allow some or all of the overland flow to infiltrate into the soil, and this further reduces the delivery of sediment to the stream network. Vegetation filter strips are usually effective unless the ground is near saturation, such as during spring snowmelt, or an extreme precipitation event generates so much overland flow that it overwhelms the filtering capacity of the buffer strip. Ephemeral channels can be an important source of sediment if these channels are not protected by buffer zones.

While soil erosion is important from the standpoint of site productivity, it is the delivery of sediment to the stream channel that is of greatest concern and the focus of most regulatory efforts. Total suspended sediment (TSS) is the concentration of solid particles in the water column, and this is usually expressed as milligrams per liter (mg L^{-1}). Turbidity is an optical measurement of the water's ability to diffract light, and is measured in Nephelometric Turbidity Units (NTU) (Stednick, 1991). Both turbidity and TSS vary tremendously over time with streamflow. Water quality standards in some states limit either the absolute amount or relative increase in these parameters. In Colorado there are no state standards for turbidity or TSS, but the state has an embeddedness standard (embeddedness refers to the degree to which a large particle is buried by finer particles). The embeddedness standard is related to macroinvertebrate productivity in a reference reach, but specific criteria and definitions have not been defined or adopted in Colorado.

Suspended sediment also can be a concern because fine particles have large surface areas per unit mass. These surface areas are reactive and may adsorb and absorb various water quality constituents, including phosphorus, introduced chemicals, and petroleum products. Because phosphorus has a low solubility in water, phosphorus exports are often correlated with the amount of suspended sediment transport. Hence the delivery and deposition of suspended sediment can affect aquatic resources both physically and chemically.

An increase in the amount of coarser particles in the stream channel is another potential concern. Bedload is defined as the transport of large particles by rolling or bouncing along the streambed. Bedload movement is difficult to measure and is not used as a water quality standard (MacDonald et al., 1991). An increase in bedload may be a major concern from the standpoint of aquatic habitat, reservoir sedimentation, channel morphology, and channel stability. The difficulty of directly measuring sediment loads means that changes in sediment inputs may be more readily detected by monitoring the physical features of the channel – such as pool volumes, amount of bank erosion, and bed material particle size – rather than direct measurements (State of Idaho, 1987; MacDonald et al., 1991; MacDonald, 1993).

The amount of suspended sediment and bedload in streams is largely governed by the characteristics of the drainage basin, and these include the geology, vegetation, precipitation, and topography, and land use. In-channel sediment sources due to scour or bank erosion can be important and should be distinguished from hillslope sources because all of the eroded material is delivered or accessible to the stream.

To achieve stream stability, a longer-term equilibrium must be sustained between the amount of sediment entering the stream and the amount of sediment being transported through the channel. Landuse activities that significantly change the amount of runoff or sediment can upset this balance and result in unwanted physical and biological changes (State of Idaho, 1987). Increases in the frequency of high flows usually increase sediment transport rates, and an increase in the size or duration of peakflows can decrease channel stability, increase turbidity and increase sediment concentrations (Brown et al., 1974; Troendle and Olsen, 1994).

The storage and routing of sediment is critical to the generation and persistence of downstream effects. Large amounts of sediment can be stored in the channel or on floodplains, terraces, and alluvial fans, and most studies show a large decrease in unit area sediment yields with increasing basin size (Walling, 1983). Larger sediment particles do not travel as rapidly and are typically more persistent in the channel network than finer particles (NCASI, 1999a). However, the storage and routing of sediment is highly variable in time and space, and this makes it difficult to predict or quantify the changes in sediment loads as a result of land use activities (NCASI, 1999a). Most studies suggest that suspended sediment concentrations and sediment yields decrease as a negative exponential after site disturbance (Leaf, 1974; Beschta, 1978; Ketcheson and Megahan, 1996). The temporal patterns in sediment production and delivery should be considered in land use planning (Stednick, 1987).

The forest practices with the greatest potential for causing erosion and stream sedimentation are road construction and intensive site preparation. Careful planning and implementation of forest practices can minimize adverse effects on water quality.

3.2.1. Roads
Numerous studies have identified unpaved roads as a major source of sediment in forested watersheds (Elliott, 2000). Specific sources include the road tread, cutslope, inside ditch, sidecast or fill material, and areas subjected to concentrated road drainage (Elliot, 2000). The highest erosion rates typically occur during road construction, and road erosion rates generally increase with road maintenance and the amount and type of traffic. Recre-

ation activities such as off-highway vehicles, mountain bikes, or trail use by foot or animal can compact the soil surface and increase soil erosion rates (Leung and Marion, 1996). Erosion rates usually decrease after construction as the disturbed sites develop an armored surface or are revegetated. Erosion rates can decline by 90% or more as roads age (Ketcheson and Megahan, 1996). Road surfaces, cutslopes, and inside ditches can continue to produce large amounts of sediment as long as traffic or road maintenance operations prevent revegetation or surface stabilization. Management practices such as the application of gravel can reduce road surface erosion rates by 80 percent or more (Burroughs and King, 1989). In the absence of any treatment, abandoned roads and trails can be slow to revegetate and continue to produce substantial amounts of sediment.

Any assessment of road erosion rates must also consider the effects of roads and trails on runoff, as this will greatly affect the rate of erosion and the subsequent delivery of sediment to the stream channel. Roads, skid trails, and other travelways have lower infiltration rates and generate more runoff than adjacent undisturbed watershed areas. The increased runoff and the construction of cut and fill slopes can generate mass movements, and in some environments this is a greater concern than road-induced surface erosion.

Road erosion rates have been measured in a variety of environments (Elliot, 2000). Table 3.1 lists published erosion rates from the road tread, cutslope, and fill slopes in the original units and in kilograms per square meter per year in order to facilitate comparisons. This table shows that road tread erosion rates can exceed 100 tons ac^{-1} yr^{-1} (22 kg m^{-2} yr^{-1}), while more "typical" values are in the range of 1-10 tons ac^{-1} yr^{-1} (0.2-2 kg m^{-2} yr^{-1}) (Table 3.1). The point is that there is a wide range of values, and some of the data from other countries, such as Australia, may be more similar to some parts of Colorado than data from areas closer to Colorado, such as Oregon or Washington.

The other key issue is how much of the sediment generated from roads gets delivered to wetlands, lakes, or the stream network. Relatively few studies have measured the delivery of road-derived sediment to the drainage network. Roads located close to streams may have a greater impact on water quality as the shorter travel distance increases the likelihood that sediment-laden waters will reach streams or other water bodies. Several studies have shown that road ditches and concentrated road drainage increase drainage density (defined as the total length of streams per unit area) (Wemple et al., 1992; Montgomery, 1995). This increase in drainage density is presumed to increase the size of peak flows and the proportion of road-derived sediment that is delivered to the stream network (Wemple et al., 1992; Montgomery, 1995). As in the case of forest management, vegetative filter strips can reduce the delivery of road runoff and erosion to surface waters (Campbell, 1984).

Sediment yields from weir pond accumulations in the Fraser Experimental Forest indicate that roads and timber harvest increased sediment yields by only 0.014 tons ac^{-1} yr^{-1} (Stottlemeyer, 1987). The increase in sediment production was attributed primarily to the increase in flow and corresponding increase in sediment transport capacity than erosion from the roads and harvest units (Troendle and King, 1987; Troendle and Olsen, 1994). Roads and timber harvest on the Fool Creek experimental watershed increased average sediment yields by only 0.005 tons ac^{-1} yr^{-1} (Leaf, 1974). Approximately 10 percent of the increase in sediment yield was attributed to flow changes and 90 percent of the sediment was related to roads. The observed increase in sediment yields declined sharply in the first 10 years after harvest.

An increase in erosion rates and sediment yields was difficult to detect on the larger-scale Coon Creek study. This was due to the limited amount of pre-treatment data and the absence of any sediment measurements except instream transport rates adjacent to the downstream gauging stations (Troendle et al., 2001). The data did not show any significant difference in suspended sediment concentrations between the treated and the control watersheds, or in the treated catchment before and after the management activities. Bedload transport rates in the treated watershed were significantly greater than for the control watershed, but an assessment of change relative to the pre-treatment condition was precluded by the absence of pre-treatment data. The greater bedload transport rates in the treated watershed are consistent with the observed increase in discharge, implying that the increase in sediment production from roads and harvest activities units were not as significant at the watershed scale as the increase in transport capacity (Troendle et al., 2001).

As noted earlier, road maintenance and high traffic volumes can increase erosion rates from unpaved roads. Road maintenance can include cleaning ditches and trimming the cut bank as well as regrading the road surface. Maintenance can generate more erosion than road usage (Luce and Black, 1999). Older roads often are of greatest concern because they tend to concentrate rather than disperse runoff, and the runoff is often directed into

Table 3.1. Published road surface erosion rates in the original units and converted to kilograms per square meter per year. One kilogram per square meter is approximately 4.5 tons per acre.

Reference	Location	Traffic, slope	Erosion Rate Reported	Sed. Product. (kg m^{-2} yr)
Road Tread				
Hoover (1945)	North Carolina	Skid trail, 30%	339 m^3/ha	203
Lieberman and Hoover (1948)	North Carolina	Access road	0.4 m^3/m	114
Hoover (1952)	North Carolina	Access road	4740 m^3/ha yr	711
Weitzman and Trimble (1952)	West Virginia	Skid road	0.048 m^3/m 0.85 m^3/m	24 43
Megahan and Kidd (1972)	Idaho	Variable use, slopes	51.0 t/mi^2 yr	7.3
Megahan (1975)	Idaho		20 t/ha yr	2.0
Wald (1975)	Washington	Moderate traffic, 6.4% Low traffic, 3.0%	44.2 t/mile yr 3.4 t/mile yr	6.6 0.48
Dissmeyer (1976)	Southeastern U.S.		8-120 t/ha yr	0.8-12
Simons et al. (1978)	North Carolina		37 t/ha yr	3.7
Buckhouse and Gaither (1982)	Oregon		0-7 t/ha yr	0.0-0.7
Reid and Dunne (1984)	Washington	Heavy traffic, 10% Moderate traffic, 10% Light traffic, 10% Abandoned, 10%	500 tonnes/km yr 42 tonnes/km yr 3.8 tonnes/km yr 0.51 tonnes/km yr	100 8.5 0.77 0.10
Swift (1984)	North Carolina	Variable use, 5-7%	0.01-1.77 t/ac mo	0.03-5.2
Bilby (1985)	Washington	Mostly high use, 1%	0.0052 t/m^2 yr	5.2
Vincent (1979)	Idaho	Light use, 6.3-13.4%	9.3-31.0 t/ac yr	2.3-7.6
Fahey and Coker (1989)	New Zealand	Light traffic, 4%		1.6-11
Froehlich (1991)	Poland	Variable slopes	0.013 m^3/m yr	9.8
Grayson et al. (1993)	Australia	Low-high use, 12-18%	50-90 t/ha yr	5.0-9.0
MacDonald and Ramos-Scharron (2000)	St. John, USVI	Heavy traffic, 12.5% Light traffic, 20.9% Light traffic, 8.3% Light traffic, 4.3%		28 24 3.8 4.3
MacDonald et al. (2001)	St. John, USVI	Light traffic, 7-18% (plots) Light-mod traffic, 0.7-14% (road segments)		0.08-2.7 0.046-7.4
Cutslopes				
Diseker and Richardson (1962)	Georgia	Unveg., NW facing Unveg., SE facing	230 t/ha 2yr 102 t/ha 2yr	12 5.1
Wilson (1963)	Oregon	6-7 yr old cutslopes new cutslopes	153 t/ha yr 370 t/ha yr	15 37
Dyrness (1970, 1975)	Oregon	5 yr old cutslopes 1 yr old cutslopes	0.5 cm/yr 0.7 cm/yr	7.5 11
Megahan (1980)	Idaho	45 yr old cutslopes, soil 45 yr old cutslopes, granite	0.01 m^3/m^2 yr 0.011 m^3/m^2 yr	15 17

Table 3.1, continued. Published road surface erosion rates in the original units and converted to kilograms per square meter per year.

Reference	Location	Traffic, slope	Erosion Rate Reported	Sed. Product. (kg m^{-2} yr)
Blong and Humphreys (1982)	Papua, New Guinea		70.0 mm/yr	100
Megahan et al. (1983)	Idaho		11 mm/yr	16
Riley (1988)	New South Wales, Australia		2.4-3.9 mm/yr	3.6-5.8
Fahey and Coker (1989, 1992) Smith and Fenton (1993)	New Zealand	Unvegetated, granite		5.2-15
Megahan et al. (2001)	Idaho	Cover density 0.1-89% 55-104% gradient	0.1-247.6 t/ha yr	0.01-25
Fillslopes				
Bethlahmy & Kidd (1966)	Idaho	Unvegetated fillslope	94 t/ha 10.5 mo	11
Megahan (1978)	Idaho	12 yr old fillslope	12 t/ha yr	1.2
Fahey and Coker (1989, 1992) Smith and Fenton (1993)	New Zealand	55% gradient, variable cover density		0.1-1.2

the stream channels. Many older roads are located immediately adjacent to stream channels or other water bodies, and this increases the likelihood that road runoff and erosion will reach the stream channels. Roads impinging on streams can also increase bed and bank erosion by reducing channel sinuosity (Schumm, 1971). Newer road designs include vegetative filter strips, more frequent drainage, outsloping of the road surface to disperse road runoff, and narrower road surfaces to reduce the size of the road tread, cutslopes and fillslopes. Whenever possible, roads are being placed in upslope or ridgetop positions rather than in the valley bottom, and this should substantially reduce the potential for adverse effects.

3.3. Stream Temperature

Disturbance or removal of the streamside vegetation will generally increase the amount of direct solar radiation reaching the surface waters and potentially increase water temperatures. Higher water temperatures will directly affect aquatic life, while an important indirect effect of higher water temperatures is the decreased solubility of dissolved oxygen. Surprisingly few recent studies have been published on the effects of forest management practices on water temperature (Beschta et al., 1987; Binkley and Brown, 1993a; Swank and Johnson, 1994). To the best of our knowledge, there are no published data from Colorado assessing stream temperature changes in response to forest management.

The dominant mechanism for stream temperature increases is the increased exposure of small streams to direct solar radiation. Other potential mechanisms include increased air temperatures, channel widening, soil water temperature increases, and streamflow modifications (Ice, in press). Small streams with lower flow rates and shallower flow depths are more susceptible to heating, but they also recover more rapidly (Andrus and Froehlich, 1991; Ice, in press). Maintaining streamside shade by retaining some or all of the streamside vegetation can minimize stream temperature increases. The experimental harvests at Coon Creek and Fraser Experimental Forest generally did not expose the channel to direct solar radiation, so the temperature changes induced by forest harvest are expected to be negligible (C. Troendle, Matcom Corp., pers. comm., 2003).

Most aquatic organisms have optimal temperature ranges and temperature increases of more than 2°C may alter development rates of aquatic organisms. An increase in sunlight and a corresponding increase in water temperatures will increase primary productivity and the growth rates of aquatic organisms. In colder high-elevation streams, an increase in stream temperature might be considered beneficial in terms of increasing productivity, particularly if the stream is used for recreational fishing. A comparable increase in stream temperatures in a lower-elevation stream might be considered detrimental, and this indicates the importance of defining the location of a potential change, the organisms that might be affected, and the beneficial uses of a given water body (MacDonald et al., 1991).

3.4. Nutrients

Forest management activities such as forest thinning and harvesting can alter the rates and processes of nutrient cycling. Catchment-scale studies have produced a large body of data on the effects of forest management on nutrient concentrations and total loads. The observed changes in nutrient concentrations vary substantially between locations, even when the studies are within a single physiographic region. Most recent studies in the U.S. have found only small nutrient losses following forest harvest and no significant degradation of water quality. There usually is minimal opportunity for a buildup of these nutrients in streams or lakes after timber harvest because the period of increased nutrient flux to the stream is normally very brief (Currier, 1980). Nutrient losses by leaching are usually minor compared to the nutrient losses by biomass removal (Johnson, et al., 1988; Mann et al., 1988; Clayton and Kennedy, 1985; Martin and Harr, 1989).

Streams emanating from undisturbed forested catchments generally have very low nitrate-nitrogen concentrations (0.002 - 1.0 mg L^{-1} NO_3-N) (Binkley and Brown, 1993a;b). Concentrations are low because nitrogen is rapidly taken up by plants and other organisms, and nitrate formation (nitrification) rates are slow. The low rates of nitrification in forested environments are due to the slow rates of organic matter decomposition, acid soil conditions, and bacterial allelopathy.

In general, nutrient mobility from disturbed forests follows the order of nitrogen > potassium > calcium and magnesium > phosphorus. Thus forest harvest or other disturbances such as fire will generally produce larger differences in nitrogen concentrations than other constituents.

Timber harvesting may temporarily increase nitrate-nitrogen concentrations in soil and stream waters. If vegetation is quickly reestablished, the increases in nitrate-nitrogen are short-lived and usually do not represent a threat to water quality or site productivity. At the Fraser Experimental Forest clearcutting increased the amount of ammonium and nitrate in the snowpack but decreased the amount of potassium. Nitrate concentrations increased in both shallow and deeper subsurface flows, while potassium concentrations increased in soil water and shallow subsurface flow (Reuss et al., 1997). The increase in nitrate-nitrogen concentrations was from 0.006 to 0.06 mg L^{-1}, but these values are still far below the criterion for drinking water of 10 mg L^{-1}. The increased ion flux was still apparent at the plot scale a decade after treatment (Reuss et al., 1997; Stottlemeyer and Troendle, 1999), but the increased nutrient flux was not detectable at the watershed scale. The estimated loss of nitrogen due to leaching over the first eight years after harvest was 48 kg ha^{-1}, while the atmospheric input over the same time period was approximately 59 kg ha^{-1} (Reuss et al., 1997).

The Hubbard Brook study in the northeastern U.S. is often cited as an example of the effects of timber harvesting on water quality (Likens et al., 1970). In this study the vegetation was cut and left on-site, and the hillslopes were sprayed with a general herbicide for three years to prevent any plant regeneration. Nutrient concentrations, particularly nitrate-nitrogen, increased significantly. This study helped identify nutrient cycling processes, but it does not represent the typical effects of timber harvesting on water quality.

In summary, timber harvest has only a minor effect on nutrient concentrations in surface waters. In the absence of extensive erosion and sediment delivery to the stream network, total phosphorus concentrations should show a significantly increase. Increased exports of nitrate-nitrogen are usually short-lived, and the concentrations are low relative to drinking water standards. Losses often are less than atmospheric inputs.

3.5. Fertilization

Fertilization to improve forest growth rates is uncommon in Colorado. Studies in other areas have reported widely varying effects of forest fertilization on water quality (e.g., Fredriksen et al., 1975; Stephens, 1975; Bisson et al., 1992; Binkley and Brown, 1993b; NCASI, 1999b). Most studies indicate no significant increase in ammonium-nitrogen and phosphorus concentrations in streams after fertilization. Nitrate-nitrogen concentrations may increase, but the increases are short-lived as nutrient retention by forest soils is excellent. As long as aerial applications do not put fertilizers directly into streams or other water bodies, forest fertilization should have little effect on nutrient concentrations in surface waters.

3.6. Prescribed Fire and Wildfire

There is an increasing body of literature on the effects of both prescribed and wild fires in forests. A number of studies have documented the changes in runoff, erosion, and water quality after wildfires (e.g., Tiedeman et al., 1978; Robichaud et al., 2000), but much less is known about the effects of fires on channel morphology and aquatic habitats. The effects of fires on water quality are highly variable, as the type and magnitude of change depends on the size and severity of the fire, site characteristics such as slope and soil type, vegetation type, amount and type of precipitation, and the rate and type of regrowth (Friedrich, 1951). Because there are so many different processes operating at different time scales, it is difficult to compare results from different watersheds (Krammes, 1960). The seemingly disparate results can only be resolved by understanding how each fire affects the underlying processes of runoff, erosion, and biogeochemical cycling.

By consuming organic matter, both controlled and uncontrolled fires cause some nutrient loss. These losses are due to nutrient volatilization, particularly in the case of nitrogen, and nutrient leaching. On the other hand, most fires alter the soil environment in ways that lead to increased microbial activity. The conversion of organic matter to ash and the increase in microbial activity usually results in a short-term increase in nitrogen availability and movement. Depending on the monitoring frequency and site conditions, an ammonium pulse may occur before the more commonly observed nitrate pulse. The short-lived increase in nitrogen availability may help stimulate any surviving vegetation or the establishment and regrowth of new vegetation.

Soil erosion rates may increase after fires, but the magnitude and persistence of an increase is highly variable and dependent in large part on fire severity (Tiedemann et al., 1979; Robichaud and Waldrop, 1994; Benavides-Solorio, 2003). Controlled or prescribed fires typically are designed to remove fuels or selected vegetation classes, or modify a vegetation type. Controlled burns usually do not consume all the protective duff or litter layer over large areas, while high severity wildfires can expose the mineral soil over relatively large areas. Post-fire increases in erosion can result from changes in a number of processes, and this can make it difficult to generalize or predict the effects of a given fire. Higher erosion rates after fires can result from: (1) the decrease in litter and vegetative cover; (2) changes in soil properties, including the loss of organic matter, formation of a water repellent layer at or below the soil surface, and sealing of the soil surface; (3) increased rill erosion due to the increase in overland flow; (4) channel incision due to the increase in runoff and decrease in surface roughness; and (5) mass movements. Soil erosion can reduce the amount of soil nutrients on site, but more nutrients are usually lost by burning the vegetation unless post-fire soil erosion rates are very high.

Some studies in the Colorado Front Range have found very little erosion after wildfires (Delp, 1968; Meyers, 1968), while erosion rates after the 1996 Buffalo Creek fire increased by several orders of magnitude (Moody and Martin, 2001). Morris and Moses (1987) measured large increases in surface erosion after several fires in the Colorado Front Range, while Cannon et al. (1997) documented an increase in mass movements after the Storm King fire near Grand Junction. A series of detailed studies on the June 2000 Bobcat fire just west of Fort Collins are showing that sites burned at high severity produce much more sediment than sites burned at moderate or low severity. Rainfall simulations on 1 m^2 plots indicate that sites burned at high severity produce 10-30 times as much sediment as unburned sites, and sites burned at moderate severity produced 2-6 times as much sediment as unburned and low severity sites (Benavides-Solorio and MacDonald, 2001; Benavides-Solorio, 2003). Monitoring of sediment production at the hillslope scale from the Bobcat and several other fires confirm the large differences in sediment production with fire severity (Benavides-Solorio, 2003).

Data from the Buffalo Creek, Bobcat, and other fires indicate that at least 80% of the annual erosion is driven by summer convective storms with rainfall intensities of 10 mm (0.4 inches) per hour (Moody and Martin, 2001; Benavides-Solorio, 2003; Kunze, 2003). In the case of the Buffalo Creek and Bobcat fires there were unusually large convective rainfall events in the first one or two summers after burning, but similar storms did not occur after the Hayman fire. It has been suggested that large burned areas may increase the likelihood of large convective storms, and this could be an important topic for further study.

Another important issue is the rate at which sites recover following burning. Plot-scale studies on the Buffalo Creek fire indicated that erosion rates on burned hillslopes were not significantly different from unburned hillslopes by the fourth year after burning. Data from rainfall simulations and sediment fences confirm that runoff and erosion rates six years after the 1994 Hourglass fire were not substantially different from the rates observed from unburned sites and areas that had been burned at low severity (Benavides-Solorio, 2003). However, at the larger scale channel incision and the deposition of sediment in downstream areas may be much more persistent. Hence the recovery of hillslope-scale processes may be relatively rapid due to vegetative regrowth and the breakdown of the fire-induced water repellency, while recovery of the incised channels and removal of the accumulated sediment in downstream areas may take many years.

Fewer studies have examined the effects of prescribed fires on erosion rates and water quality. Since prescribed fires do not cover as much area as the largest wildfires and usually are designed to burn at lower intensities, prescribed fires should have relatively small effects on water quality, especially if vegetative buffer strips are left along stream courses. Hillslope-scale monitoring of three prescribed fires in the northern Front Range is showing substantially lower erosion rates than following high-severity wildfires (Benavides-Solorio, 2003).

There is an increased interest in the effects of fire on drinking water quality. Both prescribed and wildfires can volatilize some of the forest organic matter. The resulting organic compounds can be in either a dissolved or undissolved form. These compounds may discolor water and chelate heavy metals – particularly iron and manganese – from the forest soils. Iron imparts an orange color to the water, which is aesthetically displeasing, and the manganese imparts a metallic taste to the water. These changes in water quality were observed in Strontia Springs Reservoir after the Buffalo Creek fire, but there is little or no documentation of these effects in the scientific literature. The exact mechanisms or processes have not been identified. Several municipalities along the Colorado Front Range are concerned about this potential effect on water quality, and this is another potential research topic.

Chapter 4
Historic and Projected Changes in Runoff

4.1. Natural and Anthropogenic Effects on Forests and Water Yields

As discussed in Chapter 2, a change in the type or density of forest vegetation can alter the amount of runoff. Changes in forest type and density can result from natural causes, such as succession; from climate variations; from natural disturbances such as wildfires, windthrow, and insect infestations; and from anthropogenic disturbances, such as logging, grazing, or prescribed fire. In some cases it is difficult to separate anthropogenic and natural disturbances, as fire suppression will facilitate succession, and climatic changes may be due to both anthropogenic factors and natural fluctuations. The goal of this chapter is to discuss the effects of timber harvest, fire, climate change, grazing, insect infestations and other disturbances on forest cover and water yield.

The analysis of past, current, and projected effects of forest management on water yield will focus on national forest lands, as there is more information on past and current management activities for these lands than for private forestlands. From a water yield perspective, national forest lands are of primary concern because these account for 49% of forest lands in Colorado (Figure 4.1) (DNR, 2002), and the majority of these lands are in the higher elevation areas that generate most of the state's runoff (Benson and Green, 1987). Another 19% of the forests in Colorado are managed by the Bureau of Land Management, but most of these forests are in lower elevation areas with limited potential for altering water yields.

Only 28% of the forested areas in Colorado are in private ownership, with no major holdings by private timber companies (DNR, 2002). The number of private landowners who have at least one acre of forested land has increased from 46,300 in 1990 to an estimated 200,000 in 2000 (DNR, 2002). This fragmentation of private forest lands indicates that the majority of these parcels

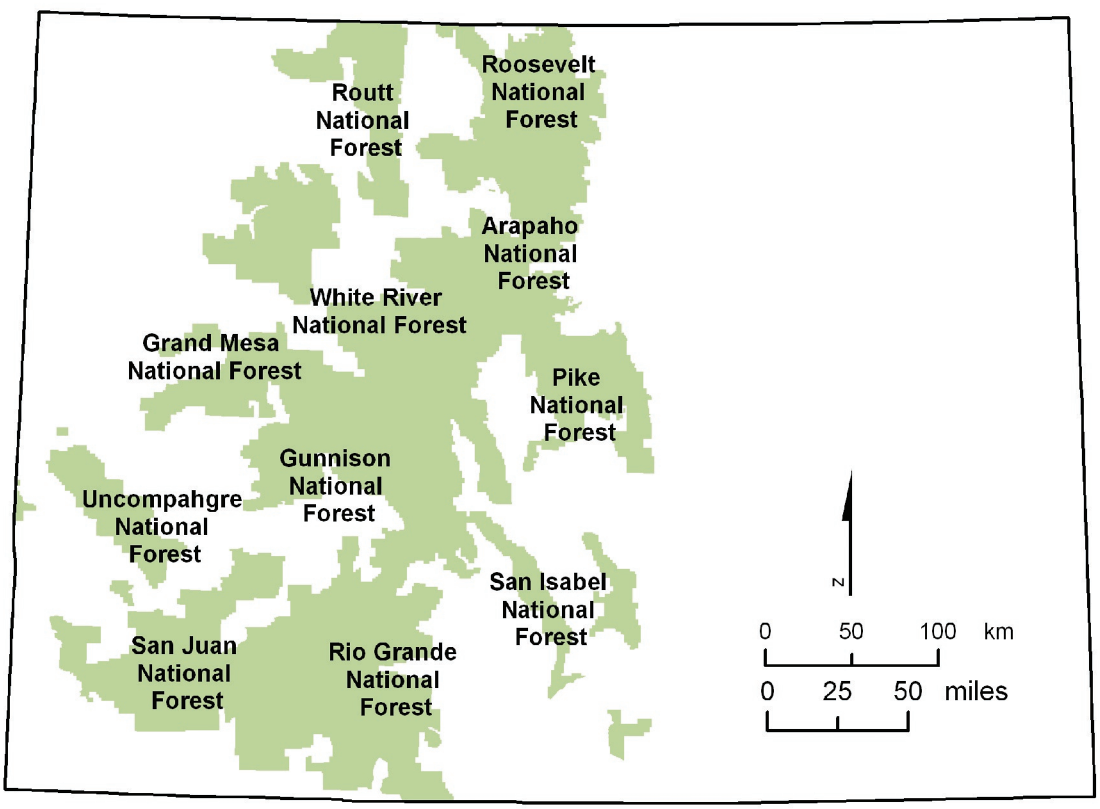


Figure 4.1. Map of national forest lands in Colorado.

are being managed for recreation, wildlife, and scenic benefits (DNR, 2002). As noted by Veblen and Lorenz (1991), the overall trend in Colorado is for forests to be used less for timber and grazing and more for recreational and residential purposes. The following sections discuss the effects of climatic fluctuations, timber harvest, fire regimes, grazing, insects and other disturbances on forests, and how these disturbances might affect past, present, and future water yields.

4.1.1. Effects of climatic fluctuations and climate changes on forests and runoff.

Changes in climate will affect the distribution, composition, and density of forests over both short and long-term time scales (Joyce and Birdsey, 2000; Veblen et al., 2000). Wetter conditions generally will lead to an increase in forest density, but this effect will occur over decades or even centuries. Increases in forest density and a shift to more mesic species can directly affect the amount of runoff. However, changes in climate can also affect forests and runoff by altering the disturbance regime (Dale et al., 2001), and the resulting changes in forest composition can have a proportionally larger effect on water yields over much shorter time scales. For example, a severe drought can increase tree mortality, which can quickly increase water yields. These increases in water yield will persist until the forest returns to its previous condition, and the time scale for this recovery is much longer than the length of the drought needed to initiate that change.

Short- and long-term changes in climate also can affect the risk of wildfires. Fire is an important source of forest disturbance in Colorado, and both short- and long-term climatic fluctuations can affect the frequency and intensity of fires by altering the frequency and intensity of drought conditions as well as the type and amount of vegetation (Keane et al., 2002). Climatic fluctuations also can affect the susceptibility of forests to beetles and other insects, and this can directly affect the amount of runoff (Love, 1955). These indirect effects of climatic fluctuations and climate change must be considered when assessing the likely response of Colorado's forests to both short- and longer-term changes in climate.

Tree-ring records and other physical evidence suggest that the treeline in Colorado shifted upwards around 1250 A.D. (Brown and Shepperd 1995). Growth rates also increased at treeline, but there is not a clear explanation for this increase in growth. The resultant increase in vegetation cover and density has probably caused a proportional reduction in water yields from higher elevation areas.

On a shorter time scale, Mast (1993) has documented an increase in the density of ponderosa pine stands in the Front Range as a result of two recent wet periods. The first wet period was from the 1870s through the 1890s, and second was from the 1970s through the 1980s. Fire suppression and reductions in grazing also may have contributed to the observed increase in the density of ponderosa pines, particularly during the second wet period (Keane et al., 2002).

This work and other data confirm that the presence and productivity of forest ecosystems in Colorado are generally limited by water rather than light (Cooper 1960). Wet periods lead to increased stand densities, particularly in more water-limited areas such as the lower and mid-elevation forests in the Colorado Front Range (Veblen et al., 2000). However, moderate droughts do not necessarily cause a corresponding increase in tree mortality. The trees established during wet periods often can survive subsequent dry periods. At least in the more fire-prone forests such as ponderosa pine, short-term fluctuations in precipitation are important because the amount of fine fuels increases rapidly during wet periods. Subsequent dry periods then have a substantially greater higher fire risk (Veblen et al., 2000), and the historic record indicates that the most widespread fires in the Front Range occurred as a result of dry periods following wet periods (Romme et al., 2002). In the more mesic, higher-elevation forests, short-term changes in precipitation directly affect the amount of runoff, but have relatively little effect on fuel loadings. Fire frequency in these more mesic forests is driven primarily by the occurrence of severe drought (Romme et al., 2002).

On a broader scale, it is increasingly recognized that humans are having a direct effect on the global climate as summarized by Intergovernmental Panel on Climate Change (IPCC, 2001). The primary cause is an increase in greenhouse gases, particularly carbon dioxide, but the increased concentration of particulates, sulfur dioxide, and other contaminants may also be affecting the climate at the regional or global scale. At this point the most clearly documented anthropogenic change on climate is a net increase in temperature. Long-term meteorological data from Colorado confirm the prediction of an overall warming trend, and this trend is due primarily to higher minimum temperatures in winter (N. Doesken, Assistant State Climatologist, pers. comm., 2001).

An increase in air temperatures due to global warming is likely to affect the direction, number, and strength of storms, and thus the amount, type and timing of precipitation in Colorado. A change in precipitation will directly affect the amount and timing of runoff, and have a longer-term feedback on runoff by altering the amount and composition of the vegetation (Joyce and Birdsey, 2000). As noted above, a change in precipitation also will have a shorter-term effect by altering the amount of fine fuels and the frequency and intensity of drought conditions (Veblen et al., 2000; Dale et al., 2001).

While it is increasingly accepted that air temperatures will continue to rise, the effects of global climate change on precipitation are much more difficult to predict. Current global climate models operate at a relatively coarse spatial scale and cannot accurately predict changes in precipitation, particularly in a topographically and climatically complex state like Colorado (Joyce and Birdsey, 2000). Unlike temperature, historic records do not show a clear trend in the amount and timing of precipitation in Colorado (N. Doesken, Assistant State Climatologist, pers. comm., 2001). Given the stronger dependence of Colorado's forests on water than temperature, it is difficult to predict the precise effect of anthropogenic climate change on the composition and distribution of forests in Colorado.

As a first step, one would postulate that an increase in air temperature will allow the further extension of forests into higher elevations and possibly increase the density and growth rate where sufficient water is available. At lower elevations and in drier sites an increase in temperature will increase potential evapotranspiration rates, and this will lead to a shift towards more xeric vegetation types and a reduction in forest cover. Current models suggest that the range of ponderosa pine would decrease due to a greater deficit of water in the spring (Joyce et al., 1990), while there are no projected changes in the range of lodgepole pine in Colorado. There are no predictions made with respect to the spruce-fir forests that produce the majority of runoff from forested lands in Colorado. The projected decrease in the range of ponderosa pine would have a minimal effect on water yields because the annual precipitation in these areas is already too low to support much of a change in water yields due to a change in vegetation type. In the absence of any other change, higher temperatures should slightly increase fire risks.

It is important to recognize that the direct effects of natural or anthropogenic climate change on runoff are likely to be much greater than the indirect effects of climate change as mediated through a change in forest cover. Chapter 2 showed that annual water yields are directly related to the amount of precipitation. An increase in temperature is an important concern in Colorado, as an increase in temperature will increase the proportion of rain relative to snow, and cause an earlier snowmelt peak. These changes could greatly affect Colorado's water supply. For example, a 5.4°F (3°C) increase in temperature has been projected to advance the timing of the snowmelt peak in the subalpine zone by 14 days and decrease the amount of summer runoff (Troendle, 1991). An increase in the amount of rain would effectively reduce the size of the winter snowpack. The combination of earlier snowmelt and more rain rather than snow will effectively increase the need for seasonal water storage, and this is an important concern for resource managers and decision-makers.

4.1.2. Historic changes in fire regimes

The different forest types in Colorado vary in fire frequency, their response to fire, and in their dependence on fires for regeneration. There also are important differences in the intensity and severity of fires in the different vegetation types. Differences in fire severity are important because high-severity, stand-replacing fires have a very different effect on the amount and timing of runoff than moderate or low-severity fires that don't kill the overstory vegetation and consume all of the protective litter layer (Benavides-Solorio and MacDonald, 2001; Benavides-Solorio, 2003).

Forest fires are an important component of forest ecosystems in the Rocky Mountains, and the different forest types show varying degrees of adaptation to fire. Forest fires facilitate the regeneration of all forest types in Colorado by exposing the mineral soil, increasing the amount of light and moisture, and releasing plant nutrients. These conditions represent an optimum seedbed for the germination and growth of the dominant forest trees in Colorado (Lotan and Critchfield, 1990; Alexander et al., 1990; Alexander and Shepperd, 1990; Hermann and Lavender, 1990; Oliver and Ryker, 1990; Perala, 1990).

Of all the forest types in Colorado, lodgepole pine (*Pinus contorta*) is the most fire-dependent. Lodgepole pines in Colorado usually have serotinous cones, so moderate- or high-intensity fires are needed to melt the resins in the cones and release the seeds. In the absence of periodic crown fires more shade-tolerant species, such as Douglas-fir (*Pseudotsuga menziesii*), subalpine fir (*Abies lasiocarpa*), and Engelmann spruce (*Picea engelmannii*), will become established and eventually replace the lodgepole pines (Lotan and Critchfield, 1990). Crown fires typically return these stands to lodgepole pine. In areas with frequent fires lodgepole pine is effectively a climax species, and in these environments it often is found in thick, "dog hair" stands that have low growth rates and high water use (Lotan and Critchfield, 1990).

In some areas with less frequent fires, lodgepole pine cones may not be serotinous and seeding may not be as fire dependent.

Ponderosa pine (*Pinus ponderosa*) forests in Colorado also are fire dependent. The periodic fires characteristic of ponderosa pine forests kill the seedlings of more shade-tolerant tree species such as Douglas-fir, and thereby sustain an overstory that can be almost exclusively ponderosa pine. In the absence of periodic fires Douglas-fir and other species can grow into the overstory and change the forest type (Oliver and Ryker 1990). In some areas aspen (*Populus tremuloides*) also can become established after a stand-replacing fire (Perala, 1990). The replacement of ponderosa pine by aspen, Douglas-fir, or other conifers may decrease annual water yields, particularly in more mesic sites with deeper soils.

Fire is much less frequent in spruce-fir forests because these species are found at higher elevations where there is more precipitation, cooler temperatures, and shorter summer dry periods. The return interval for naturally-occurring fires within the spruce-fir forest is highly variable but can be 300 years or longer (Sherriff et al., 2001; Romme et al., 2002). However, the abundance of fuels at different levels makes these forests very susceptible to intense, large-scale crown fires under exceptionally dry conditions (Keane et al., 2002; Romme et al., 2002). As noted in Chapter 2, the removal of the overstory by fire or other means will generate relatively large increases in water yields that then decay over a period of 60-70 years.

Aspen forests often grow in moister areas and do not readily burn, so fires are relatively rare. However, the thin bark means that even light surface fires can kill mature aspen. Fire-induced mortality could result in a relatively large increase in water yield per unit area. However, these increases in water yield will be shorter-lived than for other forest types because aspen grows rapidly from seed or root suckers following a fire (Jones and DeByle 1985). The rapid regrowth and thicker undergrowth characteristic of aspen stands suggests that post-fire increases in erosion will generally be less than in other forest types.

The overall trends in fire frequency and severity for the different forest types in Colorado are increasingly well documented (e.g., Keane et al. 2002; Romme et al., 2002). However, there still remains considerable uncertainty over the precise frequency, severity, and spatial extent of fires prior to European settlement. This uncertainty is least for the spruce-fir and lodgepole forests, where large fires are infrequent but usually stand-replacing, and for the lower montane ponderosa pine forests, where there were frequent, low severity fires (Romme et al., 2002). The pre-settlement fire regime is most uncertain for the ponderosa pine and mixed conifer forests in the upper montane zone, where the forests were apparently subject to frequent low-severity fires as well as stand-replacing fires (Kaufmann et al., 2000a; Romme et al., 2002).

Historic fire data are usually derived from fire scar dendrochronology, but this approach cannot determine whether a fire was natural or human-induced. Native Americans in the Rocky Mountain region used fire for a variety of purposes, including land clearing, hunting, wildlife habitat improvement, defense, and signals (Cooper, 1960; Gruell, 1985; Kay, 1995). While these fires probably differ from lightning fires in terms of their seasonality, frequency, intensity, and pattern of ignition, the fire frequencies determined from tree rings record is an inseparable mix of natural fires and the fires caused by Native Americans (Keane et al., 2002).

Most fire researchers divide the historic record into three periods, and there are prior to European settlement (pre-1850), the settlement period (approximately 1850-1920), and the suppression or fire exclusion period (1920 to present). The discussion of changes in the historic fire regime will focus on the ponderosa pine zone in the Front Range, as this is the forest type where there is the most data and where many of the most recent forest fires have occurred. The lower and mid-elevation forests in the Front Range also have been more heavily altered by past practices than most other forest types. The possible effects of fire on water yield is discussed at the end of this section, and the effects of thinning or other fuel treatments is discussed in the following section on timber harvest.

Fire recurrence intervals for ponderosa pine stands in the Front Range are believed to be longer than for ponderosa pine stands in the southwestern U.S. due to the poor growing conditions causing lower productivity and smaller amounts of fine fuels (Kaufmann et al., 2000b). However, the frequency of crown fires may be greater in the Front Range than in the southwestern U.S. (Kaufmann, 2000). Kaufmann et al. (2000b) suggested that fire recurrence intervals ranged from less than 20 years at lower elevations to around 50 years at higher elevations, while Veblen et al. (2000) estimated the pre-European fire recurrence interval for single points in the ponderosa pine zone in the Front Range to be less than 10 years. Around the Cheesman Reservoir southwest of

Denver, the mean fire interval was 50 years for individual 0.5 to 2.0 km^2 portions of the landscape (Kaufmann et al., 2001). Pre-settlement fire frequencies were lower in the higher elevation ponderosa pine and mixed conifer forests than the lower elevation ponderosa pine forests, but a higher proportion of the higher elevation fires were stand-replacing (Veblen et al., 2000). The historic fire interval for ponderosa pine forests in the southern part of Colorado is approximately 10 years and these fires were usually lower-intensity surface fires (M. Kaufmann, USDA Forest Service Rocky Mountain Research Station, pers. comm., 2001).

The European settlement period is characterized by more frequent large fires in the lower and mid-elevation forests relative to the pre-settlement period (Laven et al., 1980; Romme et al., 2002). For example, a study in Fourmile Canyon west of Boulder showed that the mean fire interval decreased from 22 years during the pre-settlement period to 7.4 years during the settlement period (Goldblum and Veblen, 1992).

After about 1900 the frequency of fires in ponderosa and mixed conifer forests in the Colorado Front Range decreased substantially. Much of the initial decline in fire frequency has been attributed to the removal of fine fuels by heavy grazing and the frequent fires during the settlement period, along with the cessation of widespread burning by the early settlers (Veblen et al., 2000; Romme et al., 2002). Fire suppression was much more effective after the severe fires in the western and northern U.S. in the early 1900s. The decrease in fire frequency has increased the amount of fuels in the lower and middle elevation montane forests. Fire suppression probably has had very little effect on fuel loadings in the upper elevation spruce-fir forests because the pre-settlement fire recurrence interval is so long relative to the period of effective fire suppression (Romme et al., 2002).

The increased forest density and changes in forest structure in the lower and mid-elevation ponderosa and mixed conifer forests has increased the risk of high severity fires. The historic fire record shows that large, stand-replacing fires were most likely to occur when several wet years were followed by an exceptionally dry period, as the wet years increased the amount of fine fuels and the dry period created the conditions suitable for burning (Romme et al., 2002). This observation supports the contention that the long period of fire suppression has increased the risk of large, high-severity fires under dry conditions by increasing the fuel loads and altering stand structure. The view that large fires are caused by the combination of high fuel loadings plus severe drought is substantiated by the large increase in the number of acres burned as Colorado and adjacent areas were subjected to an increasingly severe drought from the late 1990s through 2002 (Figure 4.2).

A recent review of the 137,000 acre Hayman fire states that fires of similar size have occurred within the historic record (Graham, in press). In particular, fires were widespread in the Front Range and much of the western U.S. in 1631 and 1851. However, the unique feature of the Hayman fire was the size and consistency of the high severity (i.e., stand-replacing) areas. Intensive studies around Cheesman Reservoir have shown that the largest area of complete mortality within the last 500 years was less than one square mile, while the Hayman fire killed virtually all of the trees within the 8600 acre study area (Romme et al., 2002). These studies suggest that the size and homogeneity of the stand-replacing component of the Hayman fire was outside the 700-year range of historical variability as documented in the Cheesman area. The problem is that there is insufficient tree ring data to show whether comparable fires have occurred in other parts of the Colorado Front Range, and the estimated recurrence interval for the Hayman fire should be placed into a broader spatial context. As an example, a 100-year storm is by definition a very rare event at a single location, but in a large area a 100-year storm could occur at some location almost every year, and may no longer be considered an exceptional event. A larger-scale assessment of fire size, fire severity, and fire frequency will require a more extensive database than is currently available. An incised alluvial fan exposed after the 1996 Buffalo Creek fire indicates that very large fire-flood events have occurred approximately every thousand years (Elliott and Parker, 2001). It also should be recognized that a fire comparable in size and severity to the Hayman fire would not be outside the expected range of variability in either spruce-fir or lodgepole pine forests (Romme et al., 2002).

The implication of the various studies is that the increased frequency of large fires during the settlement period would have decreased forest density and increased water yields. In contrast, the increase in forest cover over the last 100 or so years presumably has decreased water yields relative to the settlement and pre-settlement periods. The upper elevation ponderosa pine and mixed conifer forests would have been the primary source of the decrease due to fire suppression, as these areas receive more precipitation and generate more runoff than the lower-elevation forests. The increased forest density cover in the lower elevation ponderosa pine forests may be having little or no effect on annual water yields, as many of these areas receive less than 18-20 inches of annual precipitation. As noted in Chapter 2, any reduction in water yield would be much more pronounced in the wet years and much less evident in dry years.

The effects of high-severity fires on water yields may be greater than what could be obtained by timber harvest because the lack of ground cover and the post-fire soil water repellency may inhibit infiltration, particularly during summer rainstorms (Benavides-Solorio and MacDonald, 2001; Benavides-Solorio, 2003). The problem is that these fire-induced increases come primarily as increases in peak flows and exhibit extremely poor water quality (Chapter 3). After approximately 3-5 years infiltration rates should approach the values prior to burning, and after this point the fire-induced increases in water yield would be limited to the higher elevation sites and be similar in magnitude to the values predicted from timber harvest.

The magnitude of the water yield increase that might result from large, high-intensity wildfires in the lodgepole and ponderosa pine forests in southern Wyoming and north-central Colorado were recently modeled by Troendle and Nankervis (2000). The modeled fire was assumed to reduce the basal area by 90% on 26,000 acres of lodgepole pine and 4,000 acres of ponderosa pine within the North Platte River basin. Model predictions were based on the change in cover, and did not account for any possible change in runoff processes, so the values were effectively identical to the values that would be expected from a comparable timber harvest. They predicted that the first-year increase in water yield from the lodgepole pine area would be 15,700 acre-feet or 7.2 inches per unit burned area, while the predicted water yield increase from the ponderosa pine stands was only 1,500 acre-feet or 4.4 inches per unit burned area. The smaller water yield increase from the ponderosa pine stand is due to the fact that water is more limited in this vegetation type, so evaporation and the residual vegetation use much of the water that was formerly being transpired by the ponderosa pines (Troendle and Nankervis 2000). We know of no other effort to model the change in water yield that might be induced by wildfires.

4.1.3. Changes in forest structure over time

Timber harvest is the primary means by which land managers manipulate forest type, density, and growth rates. At least in more mesic, higher elevation areas, forest density is directly related to water yields (Chapter 2). By examining the balance between forest growth and forest harvest one can estimate the likely trends in water yields over time. Since historic data only go back to about the 1940s, several studies have used historic photographs to indicate longer-term changes in forest cover.

Most comparisons of historic and present-day photos show an increase in canopy cover in conifer stands and an invasion of meadows by pine (Veblen and Lorenz, 1991; D. Bradford, Gunnison N.F., pers. comm., 2001;

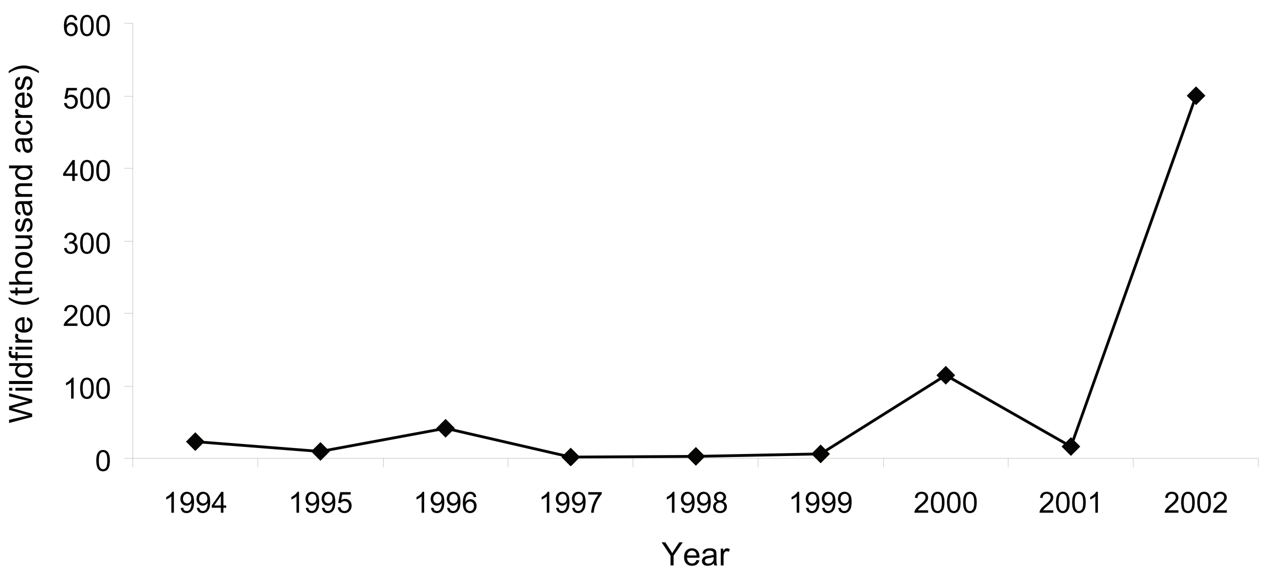


Figure 4.2. Wildfire acreage by year in Forest Service Region 2.

Keane et al., 2002). One limitation with the use of historical photos is that many of these photos date from the settlement period when the forests may already have been altered by the combination of timber harvest, more frequent anthropogenic fires, and intensive grazing. However, some of the earliest photos show the landscape before it was extensively modified by Europeans, and these also suggest that the present-day forest cover may be greater than at the beginning of the settlement period. The problem is that such photos represent only a few points, and the results cannot be reliably extrapolated to other areas, nor can the differences in forest cover be readily quantified.

A recent study did attempt to evaluate changes in stand structure and runoff from 1860 to 2000 on the 1.34 million acres of National Forest lands in the North Platte drainage basin (Troendle and Nankervis, 2000; Troendle et al., 2003). The North Platte basin straddles the Colorado-Wyoming border, and it includes most of the Medicine Bow and Routt National Forests as well as part of the Roosevelt National Forest. Existing stands were tracked backwards through time, and the changes in stand age and density were used to predict the change in runoff over time. Key limitations include the paucity of data on stand ages and the presumption that the spruce-fir stands all originated from seedlings rather than succession within an existing forest. The results indicate that stand densities have increased since 1860, with the largest increase occurring from 1900 to 1940. The best estimate of the mean annual decrease in water yield was 190,000 acre-feet or approximately 13% of the total water yield over the 1.34 million acre study area (Troendle and Nankervis, 2000). This change is effectively due to the fact that the percent of the area in pole- and saw-sized timber increased from approximately 15-30% in 1860-1900 to nearly 90% in 1980-1997. Since spruce-fir stands typically have an uneven stand-age structure, it has been argued that they should not be grown backwards in time to a seedling stage, and the spruce-fir stands should be modeled as fully occupying the sites over the entire study period. If the spruce-fir stands are assumed to be hydrologically mature for the entire study period, the decrease in annual water yields is still projected to be approximately 150,000 acre-feet (Troendle et al., 2003).

Both the analysis of historic photos and studies in the North Platte basin (Leaf, 2000; Troendle and Nankervis, 2000) indicate a general increase in forest density over the last 100-150 years. Trends over the last 50-60 years can be more rigorously assessed by using quantitative data from the eleven national forests in Colorado. Data on private timberlands are much more difficult to obtain, although there is some statewide information on the amount of land suitable for timber harvest and the amount of growing stock.

As noted earlier, forests occupy 22.6 million acres of Colorado, or one-third of the total land area (Smith et al. 1994). Only about 12.4 million acres or 55% of the forested area are considered suitable for timber harvest. National forests account for nearly 60% of the area suitable for timber harvest, while 28% is privately held. The remaining 12% of the land suitable for timber harvest is owned and managed by a variety of other federal, state, and local public agencies. There are no significant holdings of commercial timberland by the forest industry (DNR, 2002).

From 1952 to 1987 the area of forest land that is suitable for timber harvest has decreased by approximately 5 percent or 620,000 acres (Figure 4.2) (Smith et al. 1994). Most of this decrease has come from National Forest lands, and this is due to the withdrawal of lands for wilderness and other purposes, and a changing definition of where timber harvest is feasible and acceptable. An increase in the amount of BLM land that is suitable for timber harvest has partially compensated for the observed decline on National Forest lands (Smith et al., 1994). There has been little change in the amount of private land that is suitable for timber harvest (Figure 4.3).

Over this same period of 1952 to 1987 the volume of timber growing stock in Colorado has increased by slightly more than 50% (Figure 4.4) (Smith et al., 1994). This increase in growing stock can result from both an increase in forested area and an increase in the volume of growing stock per unit area as a forest matures. In Colorado it is likely that both are occurring, but the available data do not allow us to determine the relative importance of these two causes.

About two-thirds of the increase in growing stock can be attributed to national forest lands, but there also have been substantial increases in the amount of growing stock on non-industrial private lands and other public lands. In 1991 timber growth in the Rocky Mountain region was reported to be 2.6 times larger than the amount of timber being removed (Smith et al., 1994).

These data confirm that there has been an overall increase in forest density and/or forest cover in Colorado. While the causes of this increase are varied, fire suppression has clearly played a role, and another likely cause is a reduction in forest harvest for timber, fuel, and other

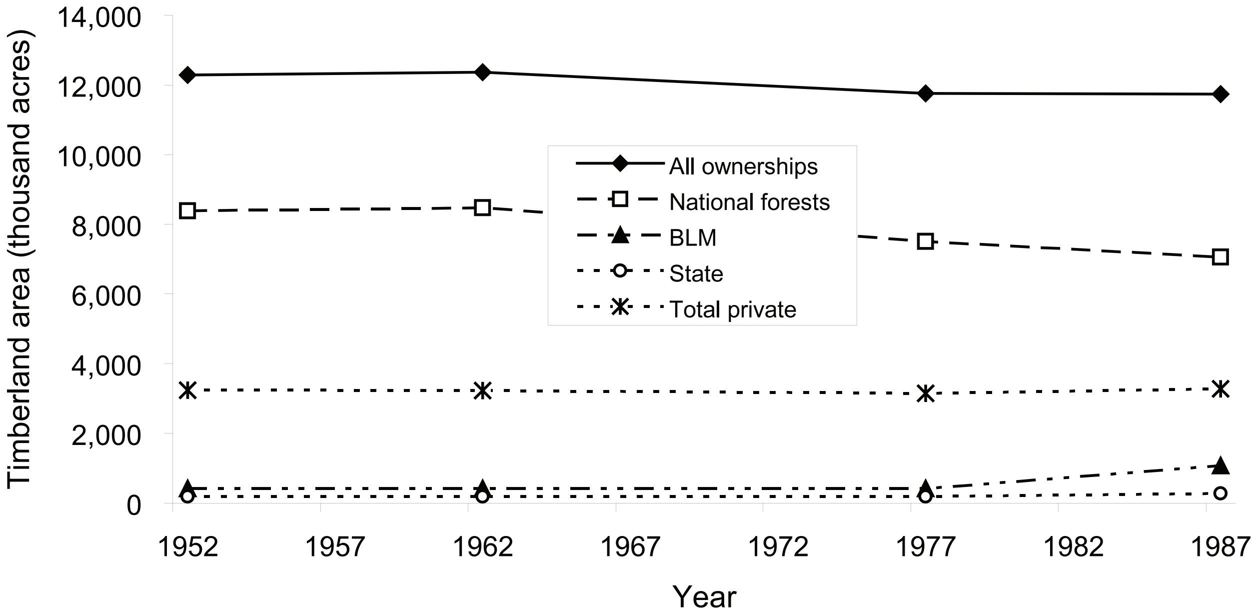


Figure 4.3. Forest land suitable for timber harvest in Colorado by ownership for 1952 through 1987 (from Smith et al., 1994).

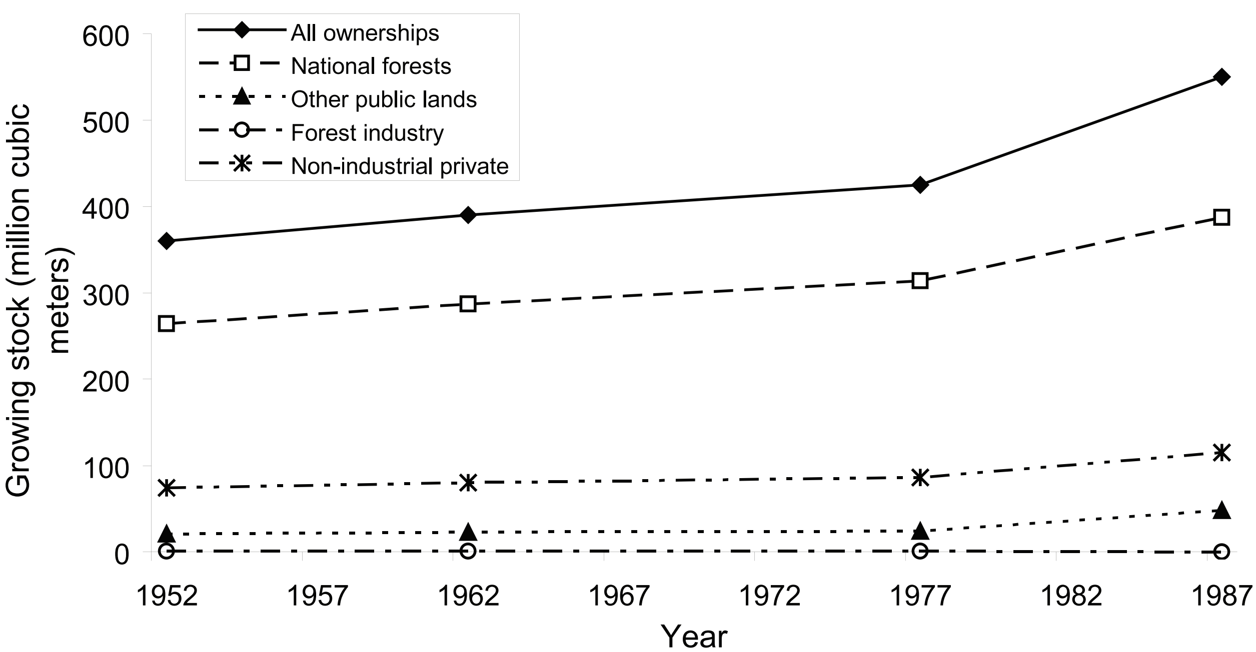


Figure 4.4. Net volume of growing stock in Colorado by ownership for 1952 through 1987 (from Smith et al., 1994).

Table 4.1. Estimated average annual water yields from national forests in Colorado. Data are from the respective forest plans. NA indicates not available.

Management Unit	Major River Basin	Annual Water Yield (million acre-feet)
Arapaho – Roosevelt	South Platte – Upper Colorado	2
Grand Mesa, Uncompahgre, and Gunnison (GMUG)	Upper Colorado	2.9
Pike – San Isabel	South Platte - Arkansas	1.3
Rio Grande	Rio Grande	NA
Routt	North Platte – Upper Colorado	NA
San Juan	Upper Colorado	2.6
White River	Upper Colorado	2.1

products. The following sections focus on timber harvest, thinning and fuel reduction activities on national forest lands, as these areas have the most complete and quantitative data.

4.1.4. Timber harvest rates on national forest lands

Timber harvest rates on national forest lands in Colorado are of interest because national forests contain approximately 60% of the land that is classified as suitable for timber harvest (DNR, 2002). Furthermore, the national forests usually occupy much of the higher-elevation areas that generate most of the State's runoff. It has been estimated that the national forest lands in Colorado are responsible for 46% percent of the total runoff in the Upper Colorado River basin, 29% percent of the flow in the Rio Grande, 7% of the flow in the Arkansas River, and 6% of the total flow in the Missouri River (Table 4.1) (Ash et al., 2000).

Because national forest lands are such an important source of runoff and they represent the majority of forests areas that are suitable for timber harvest, the management of national forest lands will have the greatest effect on the amount of forest cover in Colorado and anthropogenic changes in water yields. Data on timber harvest and thinning activities over time were compiled for each of the eleven national forests in Colorado. To the extent possible, these data are used to estimate the effects of past and current forest management activities on annual water yields. While timber harvest and other management practices also can affect water yields from private lands, harvest and stocking data are not available and private lands only account for slightly more than one-quarter of the land suitable for timber harvest. Because the changes in growing stock in each ownership category has shown the same trend as national forest lands (Figure 4.2.2), we can expect that the general trends observed on national forest lands should be similar to the trends occurring on all forest lands in Colorado.

Figure 4.4 shows the annual timber harvest from the eleven national forests in Colorado from 1940 to 2002. This indicates a rapid increase in the amount of timber harvested in about 1960, high levels of timber harvest in the late 1960s and late 1980s, and then a rapid decline in timber harvest in the 1990s. From 1960 to 1995 the average annual timber harvest from national forest lands in Colorado was nearly 170 million board-feet per year. From 1996 to 2002 the mean annual timber harvest declined to 53 million board-feet per year, or slightly less than one-third of the value from 1960-1995. While the conversion between board feet (Figure 4.5) and cubic meters (Figure 4.4) is somewhat uncertain, the average annual timber harvest on national forest lands is less than 0.02% of the growing stock.

To understand this change in harvest levels it is necessary to understand how the national forests are managed. The eleven national forests in Colorado have been amalgamated into seven management units. The overall management of each unit is governed by a land and resource management plan ("forest plan"). These forest plans are revised and updated every ten to fifteen years, and the current set of plans date from 1983 to 2002 (Table 4.2). Each plan is based on a preferred alternative that contains targets for timber sales ("allowable sale quantity", or ASQ) and other activities. Forest managers also have annual targets for timber sales and fuels treatments. Most of the forest plans include estimates for the water yield increases projected to result from the proposed management activities, as these water yield increases represent an additional product or benefit that can help determine the preferred alternative within each forest plan.

The mean annual timber sales (more precisely, the ASQ) defined in the forest plans range from 6.7 million board-feet (MMBF) for the Arapaho-Roosevelt National Forest to 36 MMBF for the GMUG and Pike-San Isabel National Forests (Table 4.2). The increase in annual water yields resulting from this level of harvest were estimated to range from 777 acre-feet per year on the Arapaho-Roosevelt National Forest to over 10,000 acre-feet per year on the GMUG and San Juan National Forests (Table 4.2). The forest plan for the Routt National Forest did not estimate the increase in water yield that would result from the proposed level of timber harvest, but we derived a rough estimate from the projected annual sale quantity. The total estimated increase in water yield from the projected timber sales is approximately 33,000 acre-feet per year. This value is considerably less than 1% of the total annual water yields from national forest lands in Colorado (Table 4.1).

When the projected water yield increases are divided by the projected annual timber sales, the resulting water yield increases range from about 87 acre-feet per million board feet on the White River National Forest to over 300 acre-feet per million board feet on the GMUG (Table 4.2).

The actual water yield increases from each management unit depend on the amount of timber that is actually harvested. Figures 4.6a and 4.6b show the annual timber harvest in millions of board-feet for each management unit from 1940 to 2002. This shows that there has been considerable variability over time and between management units. In general, the three management units on the Front Range and the White River National Forest have harvested less volume than the management units that drain predominantly to the west slope or south to the Rio Grande River.

Figure 4.7 shows the total timber harvest from all seven management units relative to the total targets from the annual plans for 1954 to 2002, as well as the actual timber harvest from 1996 to 2002 relative to the total ASQ in the forest plans. Historically, the National Forests have attained roughly 90% of the timber harvest planned by forest managers on an annual basis. On the other hand, the volumes harvested from 1996 to 2002 were only 23-48% of the volumes proposed in the current forest plans. This shortfall indicates that the harvest-induced water yield increases from national forests in Colorado are well below the values projected in the forest plans (Table 4.2). A quantitative estimate of the difference between the projected and actual changes in water yields is difficult to estimate because the harvest-induced increases in water yield depend on which areas are being harvested and the levels of harvest that have occurred in previous years.

It is important to note that, to the best of our knowledge, the forest plans only considered the effect of timber harvest on increasing water yields, and did not estimate the

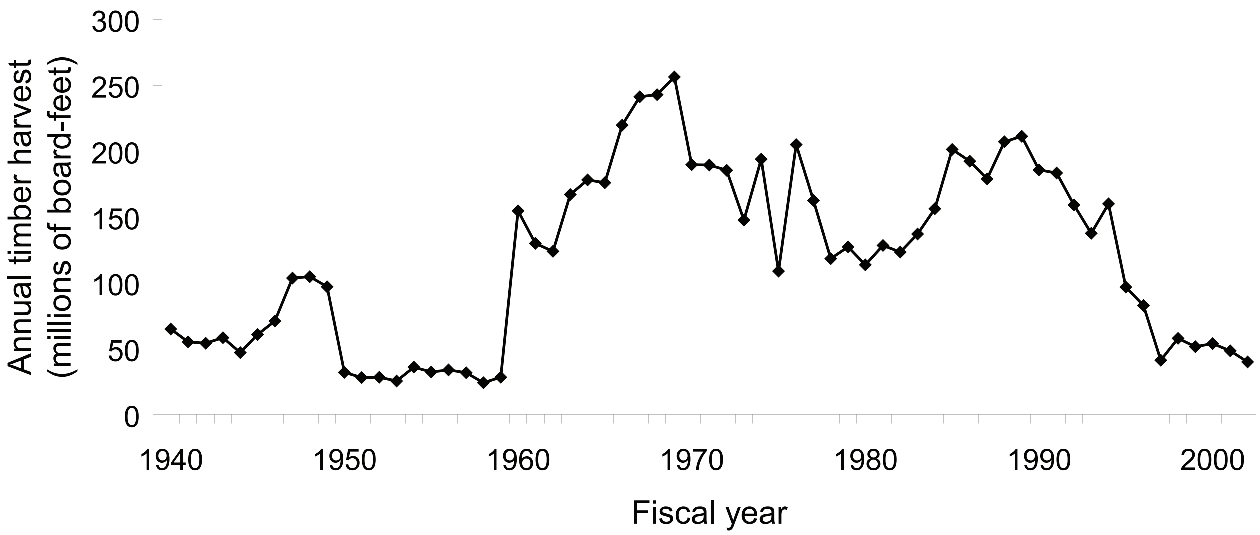


Figure 4.5. Annual timber harvest from national forest lands in Colorado from 1940 to 2002.

Table 4.2. Projected timber sales and increases in water yields for the national forests in Colorado. MMBF is millions of board-feet. Data were obtained from the preferred alternative in the most recent forest plans or from the different management units.

Management Unit	Date of Forest Plan	Projected Annual Timber Harvest (MMBF)	Projected Annual Increase in Water Yield (acre-feet/yr)
Arapaho and Roosevelt	11/97	6.5 - 6.9	780
Grand Mesa, Uncompahgre, and Gunnison (GMUG)	9/83	38.4	10,900
Pike and San Isabel	10/84	36	4,850
Rio Grande	11/96	11.3	1,500
Routt	2/98	14.8	1,850 (est.)
San Juan	9/83	24	12,000
White River	6/02	12.4	1,070

decline in water yields that may be occurring as a result of an overall change in forest cover. The first is relatively easy to calculate, while determining the changes in flow from natural growth and mortality would require a much more detailed assessment of changes in forest stands over time by precipitation zone. As an example, the simulations for the North Platte basin indicated that the increase in forest density has resulted in a net decrease in annual water yields (Troendle and Nankervis, 2000). Simulations for the period from 1997 to 2017 indicate that present water yields will decline by another 27,000 acre-feet per year due to forest regrowth. Since current levels of timber management are projected to increase water yields by only 4600 acre-feet per year, the management-induced increases in water yields will be overwhelmed by the decrease in water yields due to forest regrowth (Troendle et al., 2003).

The available data suggest that water yields from other national forests in Colorado also have declined and may continue to decline, as the amount of forest harvest is substantially less than forest growth. In other words, it is likely that the projected increases in water yields from forest harvest in Table 4.2 will be more than counterbalanced by the unquantified decreases in water yields resulting from increasing forest density and cover.

The potential for timber harvest to increase water yields was calculated in another portion of the study on the North Platte River basin (Troendle and Nankervis, 2000). This portion of the study estimated the potential increase in water yield that could be obtained from sustained timber harvesting on the 502,000 acres designated as suitable for timber harvest by the three main national forests in the study area. The management scenario assumed that all 355,000 acres of lodgepole pine and 7,000 acres of aspen would be subjected to even-aged management using a 120-year rotation. The 124,000 acres of spruce-fir would be put into uneven-aged management with a rotation age of 120 years. No water yield increase was calculated for the 3% of the area that was in ponderosa pine, as precipitation was believed to be too limiting for this vegetation type. The results showed that as each parcel was successively brought into active management by timber harvest, the average annual water yields would systematically increase until a net increase of 37,000 acre-feet per year was reached in 2015. Water yields would continue to increase beyond 2015 as more areas were brought under active management, but water yields would increase at a slower rate beyond this point because the older cuts would start to hydrologically recover. The long-term, sustainable increase in water yields for the North Platte basin was estimated to be approximately 50-55,000 acre-feet. This represents an increase of approximately 11% when averaged over the 502,000 acres of land designated as suitable for timber harvest, or 4.6% (0.5 inches) when averaged over the entire land base of 1.34 million acres (Troendle and Nankervis, 2000).

This projected increase in water yields may be too small and gradual to be statistically detectable at a typical stream gauging station, but both paired-watershed experiments and hydrologic theory indicate that this increase in flow would be present. For comparative purposes, if one assumes a volume of 15,000 board-feet per acre, the annual volume of timber that would be removed from the national forests on the North Platte River basin under this level of management would exceed the mean annual volume of timber removed from all 11 national forests in Colorado over the past six years (Figure 4.4). Troendle and Nankervis (2000) also estimated that the average annual water yield in the North Platte basin could be in-

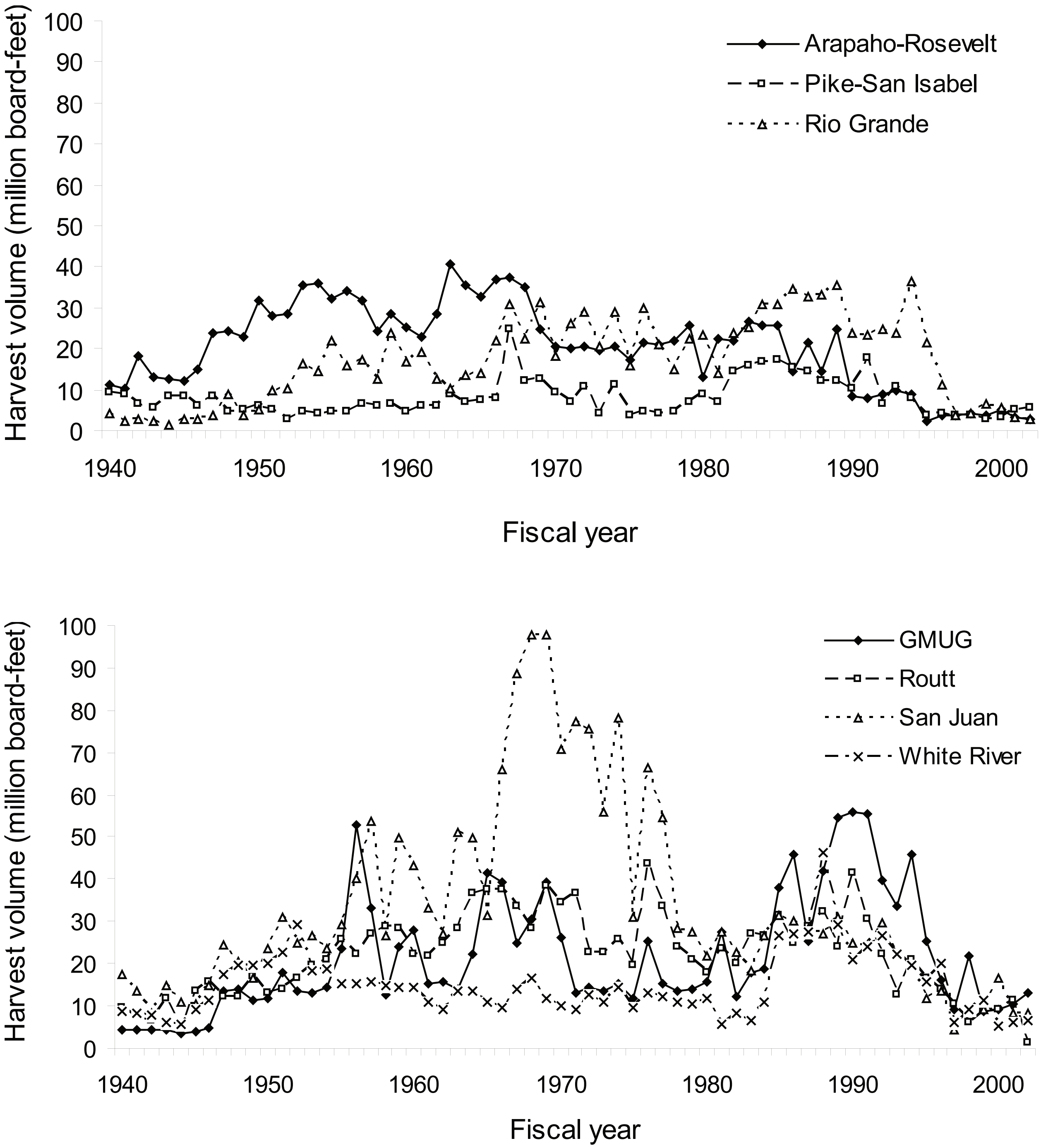


Figure 4.6. Historical annual timber harvest for: (a) predominantly Front-Range, and (b) predominantly west slope national forests. Both plots have the same scale.

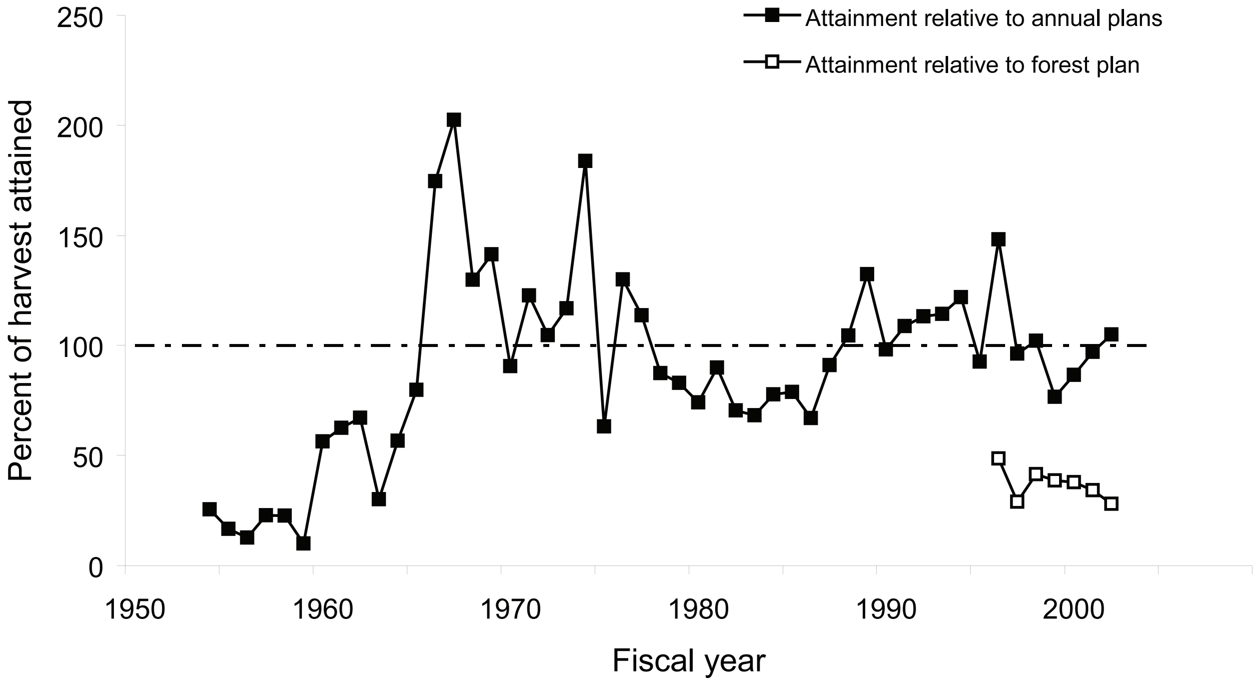


Figure 4.7. Aggregated harvest attainment levels relative to forest plans and annual plans. Dot-dash line indicates 100% attainment.

creased by another 16,000 acre-feet if the 227,000 acres classified as potentially suitable for timber harvest were brought under active forest management.

The extent to which these increases in water yield can be extrapolated to other areas or could be realized in practice is uncertain. Troendle and Nankervis (2000) noted that the stands in the North Platte River basin had been impacted by humans and natural disturbances more than many other places in USDA Forest Service Region 2. The Coon Creek watershed was reported to be almost the only suitable site for testing the effects of forest harvest on water yields at an operational scale, despite an extensive search in the central and southern Rocky Mountains (Troendle and Nankervis, 2000). To be consistent with the research conducted at the Fraser Experimental Forest and to ensure a detectable change, the goal was to harvest approximately one-third of the 4100-acre Coon Creek watershed. However, a variety of resource constraints imposed by the forest plan and technical considerations meant that only 24% of the watershed could be subjected to road construction and timber harvest.

In their study of water yields from the North Platte River basin, Troendle and Nankervis (2000, p. 21) note that mandates for multiple use and ecosystem sustainability "imply an even further reduction in the percentage of 'suitable and treatable' land base that could be dedicated to water yield augmentation." More broadly, they state that "extensive land areas suitable for water yield augmentation are not readily available on National Forest System lands in the inland west." (Troendle and Nankervis, 2000, p. 15). Clearly the amount of timber harvest that might be conducted to increase water yields on public lands is a complex issue requiring further analyses and public input.

4.1.5. Fuel treatments and water yields from national forest lands

Fuel management activities – such as prescribed burning and mechanical thinning – potentially can increase water yields by decreasing the amount of forest cover. As noted in Chapter 2, water yield increases become progressively smaller and shorter-lived with decreasing annual precipitation. At least 15% of the basal area has to be removed in order to obtain a detectable increase in water yield (Stednick, 1996). These principles suggest that water yield increases due to thinning and fuel management activities will be quite small unless extensive and heavy thinning is conducted in the wetter, higher elevation forests.

As in the case of timber harvest, data on the amount of thinning and other fuel reduction practices are only available for national forest lands. Most of the forest plans specify acreage targets for fuels treatments. Current targets for the different management units range from approximately 3000 acres/yr for the Pike-San Isabel National Forest to 9000 acres/yr for the San Juan National Forest (Table 4.3). Neither the Routt nor the Rio Grande National Forests have specific targets for fuels treatments in their forest plans.

Four of the seven management units also estimated the likely increases in water yield that would result from their fuels treatments (Table 4.3). These values range from 800 acre-feet/yr on the Pike-San Isabel to nearly 2500 acre-feet/yr for the White River National Forest (Table 4.3). The Rio Grande, Routt, and San Juan National Forests did not estimate the increase in water yields that would result from their proposed fuels treatments.

Each management unit also sets specific annual targets for fuels treatments. Over the last six years the summed annual targets have been slightly lower than the sum of the targets established in the forest plans. The number of acres treated by management unit for each year from 1997 to 2002 is shown in Figure 4.8a for the predominantly Front Range units and in Figure 4.8b for the units that predominantly drain to the west slope and the Rio Grande.

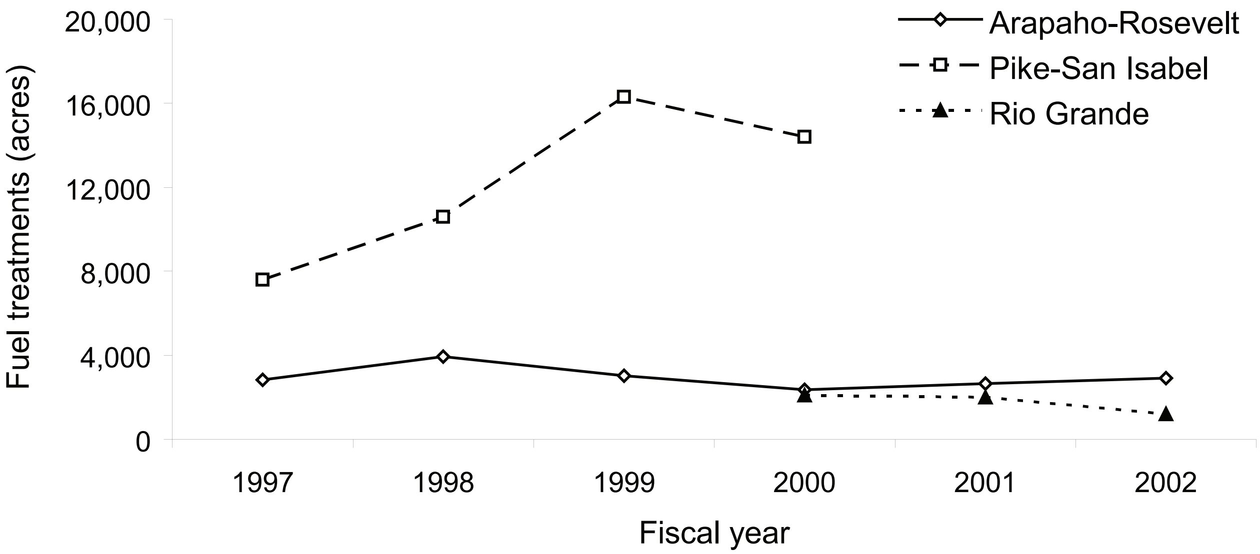
These data show a wide variation in the number of acres treated by each unit, with the San Juan and the Pike-San Isabel National Forests each treating substantially more acres than the Arapaho-Roosevelt, Rio Grande, and Routt National Forests. To some extent, these differences parallel the fire risk and extent of the wildland-urban interface in each management unit. The low number of acres treated in 2002 probably reflect the difficulty of conducting fuels treatments during severe droughts and the diversion of funds and personnel to wildfires.

From 1997 through 2000 the number of acres that were treated met or exceeded the values specified in the forest plan. The number of treated acres also exceeded the targets set in the annual plans by 20-40% (Figure 4.9). In fiscal years 2001 and 2002 the number of acres treated were only about 30-50% of the values specified in the annual plans and the forest plans (Figure 4.9). This reduction in fuels treatment attainment can be attributed to the diversion of resources to combating large wildfires as well as the ongoing drought in Colorado and accompanying difficulty in conducting prescribed burns.

These data and discussions with fuels managers indicate that the acreages treated will tend to vary between wet and dry years. When burning conditions are favorable, fuel treatment targets are generally exceeded. Under drought conditions the number of treated acres generally will fall below the target levels, and this shortfall is greatest when large wildfires require more resources. Because the national forests are generally meeting the targets for fuels treatments as specified in the forest plans, they generally should be achieving the projected water yield increases (it is beyond the scope of this report to evaluate the accuracy of these projected water yield increases). The biggest shortfalls will be in dry years when the number of treated acres may be reduced

Table 4.3. Annual targets for fuels treatments as specified in the forest plans. Projected increases in water yield assume 100% attainment. NA indicates not available.

Forest Plan Management Units	Forest Plan Date	Projected Annual Fuel Treatment (acres)	Projected Annual Increase in Water Yield (acre-feet)
Arapaho and Roosevelt	11/97	6000	1614
Grand Mesa, Uncompahgre, and Gunnison (GMUG)	9/83	6000	1600
Pike and San Isabel	10/84	3000	800
Rio Grande	11/96	NA	NA
Routt	2/98	NA	NA
San Juan	9/83	9000	NA
White River	4/02	6100	2480



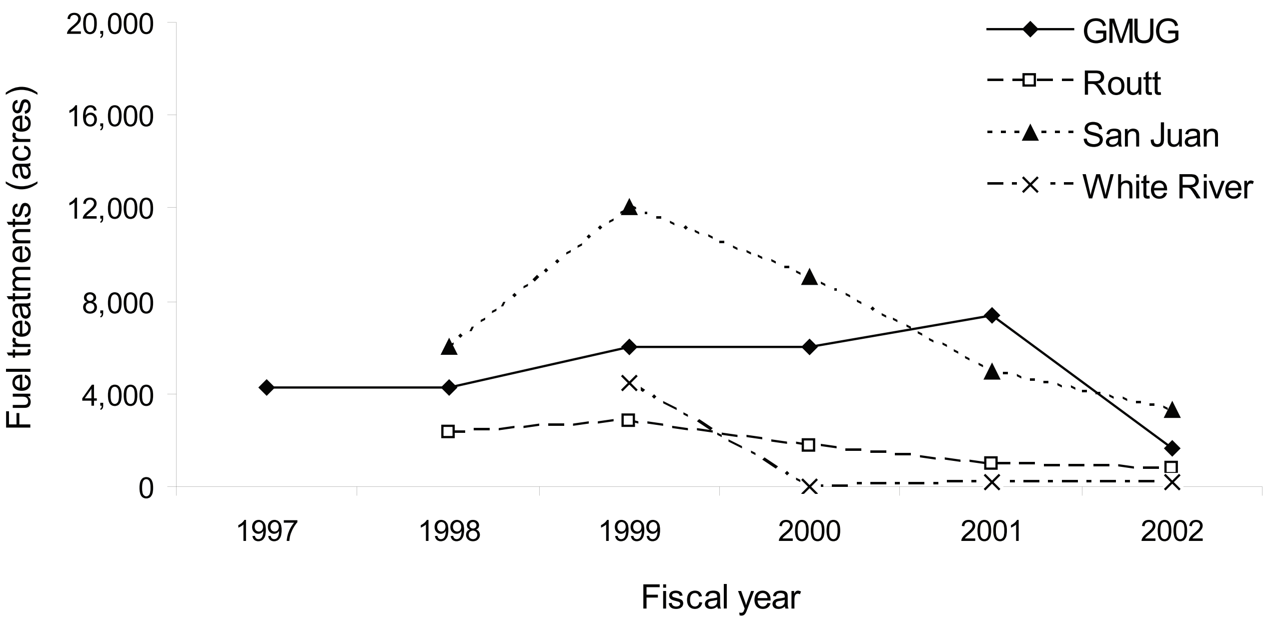


Figure 4.8. Acreage of fuel treatments by year for the National Forests that predominantly drain (a) to the Front-Range, and (b) to the west slope of the Continental Divide and the Rio Grande.

and there is less precipitation. Recent wildfires are causing a shift in emphasis towards more fuels treatments, and this should result in more acres being treated and a corresponding increase in projected water yields.

4.1.6. Grazing

Domestic grazing in Colorado began in the mid-to-late 1800s (Cooper, 1960). At first sheep were dominant, but cattle became more prevalent over time. The direct effect of grazing is that this has allowed the encroachment of conifers into meadows and other areas (Covington and Moore, 1994). Grazing within existing forests generally is not believed to affect forest density and composition except in the case of aspen forests. Aspen forests produce more forage than most other forest types, and these areas can be more heavily impacted by wild and domestic grazing (DeByle, 1985). Grazing reduces the regeneration of aspen, and this can help convert aspen stands to coniferous forests or other vegetation types.

Intensive grazing reduces fine herbaceous fuels, especially grasses, and this can reduce the frequency and extent of low-intensity surface fires. The decrease in low-intensity fires in the early 1900's has been attributed to the reduction in fine surface fuels from grazing (Savage, 1989). Mast (1993) attributes the large increase in pine seedlings in the 1970s and 1980s to the combination of a decrease in grazing and higher than average precipitation. A reduction in low-intensity surface fires increases the risk of high-intensity fires in southwestern ponderosa pine forests (Covington and Moore, 1994; Allen, 2002), but it is not clear if grazing has the same effect in the Colorado Front Range. Grazing also can increase the risk of high-severity wildfires by facilitating the establishment and growth of conifers, which can increase fuel loadings. By reducing the amount of fine fuels and facilitating conifer establishment, heavy grazing can effectively exacerbate the effects of fire suppression (Gruell, 1983; Keane et al., 2002). Moderate or light grazing has much less effect on vegetation type, fuel loadings, and watershed condition.

4.1.7. Insects, disease, and other abiotic disturbances

Other processes that can affect forest density and composition include biotic factors, such as insects and diseases, and abiotic factors such as windthrow and drought. Insect infestations are an important, periodic source of forest disturbance for most forest types in Colorado. Some of the primary insects of concern include the spruce beetle (*Dendroctonus rufipennis*), western spruce budworm (*Choristoneura occidentalis*), western balsam bark beetle (*Dryocoetes confusus*), mountain pine beetle (*Dendroctonus ponderosae*), and pine engraver (*Ips pini*) (Alexander and Shepperd 1990, Oliver and Ryker 1990).

These insects are always present, but when they reach epidemic populations they can cause substantial mortality over relatively large areas. For example, nearly 6 million board-feet of timber was damaged by spruce beetles from 1939 to 1951 (Alexander and Shepperd 1990). Troendle and Nankervis (2000) modeled the effects of a beetle epidemic on water yield on spruce forests in the Routt, Roosevelt and Medicine Bow National Forests. They assumed 50% mortality in sawtimber stands on 230,000 acres over a 10-year period and 30% mortality on 40,000 acres in pole-sized stands. This assumed mortality resulted in an average water yield increase of 2.2 inches or 56,100 acre-feet by year 10, followed by a return to pre-epidemic water yields over the next 60-70 years. These model results are consistent with the observed increases in water yield following a severe spruce beetle infestation on the White River National Forest (Love 1955).

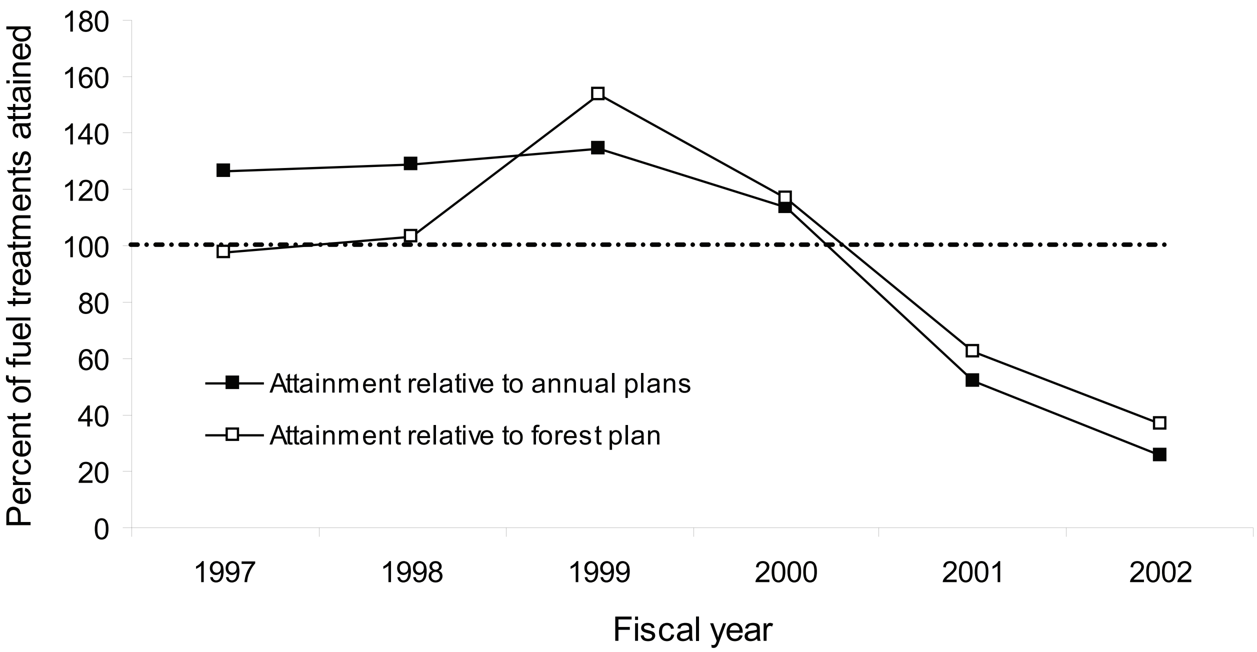


Figure 4.9. Percent of fuel treatment attained relative to forest plans and annual targets. The values shown are only for those management units where data were available. The dot-dash line represents 100% attainment.

Drought, fires, and other factors that kill trees or reduce stand vigor can trigger an increase in insect populations that can spread into other areas. Forest managers are concerned about the spruce beetles emanating from the 13,000 acres of blowdown on the Routt National Forest in 1997.

There are a variety of other pests and diseases that can affect forest health and thereby annual water yields. Dwarf mistletoe (*Arceuthobium* spp.) and various heartwood rots (e.g., *Phellinus* spp.) account for approximately 25% of the total tree mortality in Colorado (USDA, 1987). Forest diseases often occur in conjunction with pest outbreaks, resulting in pest complexes.

Abiotic factors, such as windthrow, account for 11 percent of the total tree mortality in Colorado (USDA, 1987). Windthrow can eliminate forest cover in large areas. In 1997 approximately 13,000 acres of trees were flattened by windthrow in the Routt National Forest. For comparison, this blowdown would convert to approximately 13 years of timber harvest assuming the proposed ASQ in the forest plan of nearly 15 million board feet and a stand volume of 15,000 board feet per acre.

The implication of these data is that insects, disease, and blowdown are all important causes of forest disturbance in Colorado. These sources of disturbance vary considerably in their periodicity and severity, but they can affect most forest types. Both modeling and field studies show that intensive disturbance in the higher elevation forests can increase annual water yields, and the changes in water yields that result from these disturbances can be much larger than the changes in water yield resulting from current timber harvest and fuel treatments.

Chapter 5
Effects of Past, Present and Future Forest Management on Water Quality

5.1. Introduction

The purpose of this chapter is to identify the likely trends in water quality that have occurred as a result of changes in forest cover and forest land management, and discuss likely changes in water quality resulting from different forest management scenarios. The discussion will draw heavily on the material presented in Chapter 3 and parts of Chapter 4. In contrast to the sections on water yield, the discussion of past and future changes in water quality will be largely qualitative, as we have less research data and weaker predictive capabilities. Nevertheless, the overall trends and effects of different management scenarios can be clearly identified, and this should help clarify the debate regarding the effects of different forest management scenarios on water quality. The gaps in knowledge and understanding identified in this chapter are presented in more detail in Chapter 6.

5.2. Historic Trends in Water Quality

In assessing past changes in water quality it is critical to specify the basis for comparison. Chapter 4 documented a general increase in forest density and cover since the settlement period of approximately 1850-1910, although the magnitude of change in the higher-elevation spruce-fir forests is much less than in the lower-elevation ponderosa pine forests. It is less clear how the current extent and composition of forests in Colorado compares to the pre-settlement period. Troendle and Nankervis (2000) concluded that, at least for the national forest lands in the North Platte River basin, present forest density is greater than at the beginning of the settlement period in the mid-1800s. In the low-elevation ponderosa pine forests in the Front Range research suggests a regime of frequent low-severity fires and relatively infrequent high-severity fires prior to European settlement (Romme et al., 2002). Since low-severity fires generally have little effect on water quality, it is reasonable to assume that Native Americans had minimal adverse effects on water quality in these lower elevation forests. The time scale between disturbance events in the higher elevation spruce-fir forests is much longer, and Native Americans probably had little effect on the quality of water emanating from these areas.

Conversely, the changes in forest cover and land use during the settlement period probably had a large and generally adverse effect on water quality (Wilkinson, 1992; Wohl, 2001). In the 19th century Colorado's forests were subjected to widespread removal of beaver, heavy grazing, and timber harvest. Streams, riparian zones, and water quality were subject to much less control than today. Splash dams were used to transport logs downstream, and this severely affected stream channels (Young et al., 1994; Wohl, 2001). Water quality and aquatic habitats were degraded by placer and gravel mining. Alluvial mountain streams and valleys were subjected to dredging, and this severely altered those channels, riparian zones, and valley bottoms. In many forested areas the legacy of beaver removal, splash damming, mine tailings, and dredging continues to affect water quality and aquatic habitat (Wohl, 2001; Fausch and Young, in press). During the settlement period there also was an increase in the number of large fires (e.g., Laven et al., 1980), and this would have adversely affected erosion rates and water quality relative to the pre-settlement period.

Since the late 1800s and early 1900s there has been a general trend towards decreasing forest harvest for fuel, timber, and other products, a reduction in forest grazing, less area being burned by high-severity wildfires, and more restrictions on in-channel activities such as placer and gravel mining. These changes probably have improved the overall quality of water emanating from forestlands in Colorado relative to the settlement period. However, current water quality is almost certainly impaired relative to the pre-settlement period due to the multiple effects of roads, settlements, the loss or degradation of riparian areas, grazing, acid mine drainage, gravel mining, placer mining, and highway sanding and salting. Since the focus of this report is on forest management, the following sections only discuss how present and future trends in timber harvest, forest grazing, fires, and climate change are likely to affect water quality.

5.3. Timber Harvest, Roads, and Grazing

Chapter 4 showed that over the past 50 years there has been an overall increase in growing stock on forestlands in Colorado. Timber harvest on national forests in Colorado peaked from 1960 to 1995, and current levels are only about one-third of the mean value from 1960-1995. A lower rate of timber harvest implies that fewer acres are being disturbed by timber harvest and site preparation. At least for public lands, the combination of lower harvest levels and increased use of Best Management Practices (BMPs) implies that timber harvest is probably having little effect on water quality. A more important issue is how changes in the rate of timber harvest affect the amount of construction, maintenance, and amount of traffic on unpaved, native surface roads. Lower levels of timber harvest should result in less road construction, less road grading, and less traffic, which should reduce the amount of sediment being produced and delivered to the stream network (Luce and Black, 1999). On the other hand, the reduction in timber harvest may be reducing the amount of funds for rehabilitating or removing the older roads that are often the largest sources of sediment.

A potentially more important trend is the increasing recreational use of forest roads. An increase in traffic generally will tend to increase erosion rates and necessitate more frequent road grading. Public land managers are attempting to minimize the impact of recreational traffic by closing roads and restricting the use of off-road vehicles, but there are few data on the amount, type, and timing of traffic on unpaved roads in forested areas, and this severely limits our ability to quantitatively assess the effects of forest roads on water quality. Similarly, there almost certainly has been a substantial increase in the number and use of roads on private lands as a result of the increasing population density in forested areas. The increases in development and recreational use are probably having a far greater impact on water quality than any decrease in disturbance and traffic due to the reduction in forest harvest. Because unpaved roads are often the dominant source of sediment in forested areas, there is a need to collect some basic data on road erosion rates, trends in road density and traffic, and the proportion of road-derived sediment that is being delivered to streams, lakes, and wetlands.

Unusual events such as the Routt blowdown or large areas of beetle kill are unlikely to have a direct, adverse effect on water quality because of the lack of roads or other ground disturbance. As noted in Chapters 2 and 3, a decrease in forest density due to timber harvest, blowdown, or beetle kill will increase the amount of runoff and the size of peak flows. However, the data presented in Chapters 2 and 4 suggests that such increases will probably be too small to trigger detectable increases in bank erosion or a significant degradation of aquatic habitat.

Grazing can adversely affect water quality by reducing vegetative cover, reducing infiltration, and contributing fecal material (Blackburn et al., 1982). Heavy grazing in the riparian zone is of particular concern because this can reduce the protective vegetation along the stream banks and greatly increase the amount of bank erosion (Trimble and Mendel, 1995; Belsky et al., 1999). These problems are generally greater in rangelands than forested areas. The overall trend on national forest lands has been to reduce the amount of grazing and better control the timing and level of use. These changes have probably led to some improvements in water quality and stream channel condition, particularly in areas where adverse impacts had been observed in the past. On some public lands, such as in Rocky Mountain National Park, overgrazing can be locally significant due to the increasing number of large ungulates and the lack of predators (Hess, 1994; Graf, 1997). Because such areas are relatively small and generally located in headwater areas, these types of localized impacts should have relatively small effects on downstream resources. The magnitude and effects of grazing on private forest lands are unknown.

With respect to the future, National Forests and other public agencies are likely to try and minimize adverse impacts by closing or relocating roads, and controlling the use of off-road vehicles. It will be much more difficult to reduce the amount of traffic on existing roads, and only limited funding is available to reconstruct or relocate the older, unpaved roads that are immediately adjacent to the stream channel and probably pose the biggest threat to water quality. On private lands continuing development is likely to increase the amount and use of unpaved roads. The large number of roads and high cost of mitigation means that unpaved roads in forested areas are likely to have a continuing adverse impact on water quality and aquatic resources.

5.4. Fire and Water Quality

Along with roads, severe wildfires generally have the greatest effect on water quality. Chapter 4 documented a higher frequency of large fires during the European settlement period (approximately 1850-1910) relative to either the pre-settlement period or the suppression period with the exception of the past several years (i.e., approximately 1910 to 1999). Since large, high-severity wildfires pose a much greater threat to water quality than low-severity fires, one would expect better water quality over much of the suppression period than during the settlement period.

The future effects of fires on water quality depends on the size, severity, and frequency of fires and subsequent rainstorms. The combination of fire suppression, reduced timber harvest, and reduced forest grazing has resulted in a denser forest cover in some forest types, and it is increasingly accepted that these changes have increased the risk of high-severity wildfires (Allen et al., 2002; Keane et al., 2002; Romme et al., 2002). In the absence of intensive efforts to reduce fuel loadings by prescribed fires or mechanical treatments (including commercial timber harvests), there will be a continuing or increasing risk of large, high-severity wildfires in the lower and mid-elevation ponderosa pine and mixed conifer forests.

With respect to water quality, a policy of continued fire suppression may provide the greatest protection in the short term. However, as observed in the past several years, areas with high fuel loadings are increasingly likely to burn in a high-severity wildfire under progressively drier conditions. Chapter 3 showed that erosion and sedimentation rates after high severity fires can increase by two or more orders of magnitude relative to undisturbed forests or areas burned at low severity. Once an area has burned at high severity, it can be very difficult to prevent the characteristically large increases in runoff and erosion, particularly for larger storm events. A recent review of emergency post-fire treatments such as seeding and contour felling stated that "the amount of protection provided by any treatment is small" (Robichaud et al., 2000). These conclusions are generally supported by recent studies on the effective of post-fire rehabilitation treatments after the Buffalo Creek (C. Clapsaddle, pers. comm., 2000) and Bobcat fires (Wagenbrenner, 2003). This means that efforts to protect water quality by continued fire suppression will be punctuated by periodic instances of severely-impaired water quality after large wildfires.

The other end of the spectrum of possible management scenarios is a widespread effort to reduce fuel loadings by reducing forest density. The reduction in fuels can be achieved by a variety of different actions, and these include prescribed fire, commercial harvest, and non-commercial thinning. If done carefully and in accordance with BMPs, and the increased traffic and road grading does not greatly increase the amount of sediment delivered to the stream network these practices should have minimal adverse effects on water quality. Such fuel treatments, while possibly having small but more chronic effects on water quality, can reduce the likelihood of crown fires (Pollet and Omi, 2002) and by implication high-severity wildfires. A reduction in the risk of large, high-severity wildfires will proportionally reduce the risk of large, post-fire increases in erosion and sedimentation. A reduction in erosion from treated areas after a subsequent wildfire has been documented for chaparral ecosystems in southern California (Wohlgemuth, 2000).

In Colorado about 10,000-45,000 acres of national forest lands are currently being treated for fuels reductions each year. This represents from 0.1 to 0.4% of the total area in national forests. A large expansion of such efforts on public lands will have to face a series of issues such as cost and public acceptance. An increase in the use of prescribed fire will face issues of air quality and the limited number of days with meteorologic conditions suitable for conducting controlled burns.

If we accept that: (1) high-severity fires can have severe adverse effects on water quality and downstream aquatic resources, (2) fuel reduction programs reduce wildfire risk while having only small effects on water quality, and (3) post-fire rehabilitation efforts have limited utility, it follows that a proactive program of fuel treatments will probably result in better water quality over the long term than continuing efforts to suppress wildfires and then applying emergency rehabilitation treatments when wildfires do occur. Since it will not be possible to immediately reduce fuel loadings on all areas within the wildland-urban interface or all areas judged to have excessive fuel loadings, it follows that a systematic procedure is urgently needed to identify which areas should have the highest priority for fuel reduction treatments.

From a public policy perspective, the highest priority areas should be those with a high wildfire risk and high densities of people and houses. Source areas for domestic water supply might have the next highest priority for treatment, followed by areas where post-fire erosion runoff and erosion would adversely affect high-value aquatic resources.

The development of an explicit ranking process for fuel treatment programs is hindered by gaps in the current state of knowledge. The prediction of wildfire risk requires a spatially-explicit database on fuel loadings and ignition events. The assessment of erosion risks from different fuel treatment procedures requires data on erosion rates from different fire severities in different vegetation types and climatic zones. These data then have to be combined with spatially-explicit information on the human and natural resources at risk. The development and integration of these assessment tools is an important research need, and this is discussed in Chapter 6.

5.5. Climate Change and Water Quality

Climate change may have a significant long-term effect on forest cover and density, but the effects are difficult to predict because of the uncertainty with respect to future changes in precipitation. Current models and long-term temperature data both indicate that winter minimum temperatures have been increasing, and this should allow some forest expansion into the alpine zone and possibly an increase in forest density near treeline. While this increase in forest cover might marginally improve water quality, the same changes in climate might decrease forest cover at lower elevations. A decrease in forest cover at lower elevations is of greater concern because these areas are subject to more intense rainstorms and hence have a greater potential for surface soil erosion.

The issue of greater concern is how climate change might affect water quality by altering the frequency and severity of large fires. In the absence of a change in precipitation, an increase in temperature will increase the likelihood of forest fires due to the increased drying and earlier snowmelt at higher elevations. A change in the amount and timing of precipitation could have an even larger effect on the frequency and location of large fires. Climate change might also affect the number and severity of large rainstorms, and this could also affect water quality by altering runoff and erosion rates. The problem is that current models do not consistently indicate whether precipitation in Colorado might increase or decrease, and historic precipitation data do not show a clear trend over time (N. Doesken, Colorado State University, pers. comm., 2001). In the absence of a reliable prediction or documented trend in precipitation, it is not possible to predict how global climate change might affect fire frequency and fire severity, and thus how a change in climate might affect water quality.

Chapter 6
Information Needs

The purpose of this chapter is to identify the primary information needs with respect to the past, present, and future effects of forest management on runoff and water quality. Most of these needs were identified in the different chapters of this report, and this chapter brings these ideas together and presents them in a more complete and integrated manner. To the extent possible, this chapter indicates the relative priority of these different information needs, and whether they need to be addressed by additional basic research or whether they can be addressed by compiling and using existing information.

6.1. Water Yields, Peak Flows, and Low Flows

Chapter 2 provided an overview of the effects of forest management on runoff. There has been a relatively large amount of research on this topic, and this allows us to accurately predict the effects of forest management on runoff in spruce-fir and lodgepole pine forests in the sub-alpine zone. Less information is available on how forest management might affect runoff in other forest types, such as the mid-elevation ponderosa pine and mixed conifer forests. A few plot-scale studies have been conducted (e.g., Gary, 1975), and current knowledge of hydrologic processes and tree physiology can be used to estimate how vegetation change in these other forest types might affect annual water yields, the size of peak flows, and the amount of low flows. However, no paired-watershed studies have been conducted in Colorado on these other forest types. On a statewide basis, the primary limitations with respect to quantifying the effects of forest management on runoff include: (1) the lack of quantitative data on changes in forest cover over time; (2) a more explicit understanding of how changes in forest cover affect all of the different flow percentiles rather than just monthly or annual water yields; (3) an integrated assessment of the opportunities and constraints for changes in forest harvest rates to increase water yields in different river basins.

With respect to the first issue, the evidence generally indicates that forest cover and forest density has increased from the late 1800s to the present. The problem is that much of this evidence consists of photo comparisons, historical accounts, or very localized studies. There is a need for a more systematic assessment of changes in forest cover and age structure in different river basins, as this information provides the basis for calculating historic changes in streamflow. For national forest lands it may be possible to quantify changes in forest cover from the early 1900s to the present by a systematic evaluation of historic stand records. An assessment of stand density and composition prior to the European settlement is much more difficult, as this probably would require detailed dendrochronology studies in different locations. Both types of data are needed to determine how current streamflows have changed with respect to the pre-settlement and settlement periods, respectively. The presumption is that present-day streamflows have declined relative to 100 or so years ago, but we know very little about how present-day streamflows compare to the estimated streamflows prior to European settlement.

An assessment of the historic changes in forest cover and the associated changes in water yields are likely to be of interest to water supply managers, but there are two reasons why these assessments may not be a high priority. First, it is questionable whether pre-settlement forest conditions and water yields should be used as the baseline for comparison. An understanding of these conditions may be important for evaluating changes in aquatic habitat and establishing management goals, but it clearly is impossible to return all of Colorado's forests to "pre-settlement conditions" and thereby re-establish the pre-settlement streamflows. It also is important to recognize the temporal and spatial variability in pre-settlement conditions due to disturbance from fires, insects, disease, and blowdown. Second, it is questionable as to whether the calculated changes in flow relative to pre-settlement conditions should guide future forest management. Funds and effort might be better spent on predicting the potential changes in runoff from different management alternatives than evaluating the changes in flow relative to past conditions.

The second information need is to more precisely determine the changes in different flow percentiles following different types of disturbances (e.g., timber harvest, fire) in different forest types. In this technique the hourly or daily flow data are simply ranked to determine the percent of time the flow is at or below a given discharge. A comparison of flow duration curves from treated and control watersheds or over time can be used to determine the change in flow for any flow percentile. This provides a more comprehensive understanding of the changes in runoff than the more common comparisons of annual water yields, the size of the annual maximum peak flow, or low flows (Austin, 1999). By separating the post-treatment period into different periods, it may be possible to quantify how different flow percentiles recover over time. This type of analysis is relatively easy to conduct, and would lead to a more precise understanding of how forest management affects the entire range of flows. The primary limitation is the lack of flow data from experimental watersheds or watersheds with long-term flow data and known changes in forest cover.

The third information need with respect to runoff is a better understanding of management constraints on increasing water yields. In evaluating the potential to increase water yields from the North Platte basin, Troendle and Nankervis (2000) assumed that the areas designated as suitable for timber harvest could be placed under active forest management. However, their report also suggests that all of the area designated as suitable for timber harvest may not be available for active management given the various technical considerations and management constraints. The balancing of different resource needs on national forests is addressed through the development of the forest plans and the selection of a preferred alternative. In the present forest plans the allowable sale quantities are much lower than the theoretical value that could be achieved if all of the area classified as suitable for timber harvest was being harvested on a sustained yield basis. To facilitate future management decisions, there is a need for decision makers and the public to be better informed about the various trade-offs associated with different levels of thinning and forest harvest in different forest types. By definition, this report has focussed on how forests affect water quantity and water quality, but there are a wide range of other social, economic, biological, and legal issues that also have to be addressed (e.g., Ponce and Meiman, 1983).

Of particular concern is the extent to which existing reservoir storage is available to capture an increase in flows due to timber harvest or other types of disturbance (e.g., insects, disease, or windthrow). Chapter 2 noted that forest harvest in the sub-alpine zone has a much greater effect on water yields than harvest in the mid- or lower elevation zones, and that nearly all of the increase in runoff comes on the rising limb of the snowmelt hydrograph. Increases in water yields also are much larger in wet years than dry years. In most cases an increase in water yields will only be useful if the additional water can be stored and used later in the year, or in subsequent dry years. The implication is that a basin-by-basin analysis must be conducted to determine the seasonal and interannual storage capacity under different operating scenarios and climatic conditions. Efforts to quantify the changes in forest cover over time and the potential for increasing water yields should focus on those basins where an increase in runoff could be captured and would be most beneficial (Brown, 1990).

6.2. Frequency of Wildfires and Effects on Runoff and Water Quality

The effect of wildfires on runoff and water quality is one of the most important issues for water managers. Chapters 3 and 5 indicated that wildfires can increase the size of peak flows and erosion rates by two or more orders of magnitude, and adversely impact water quality by increasing turbidity and the concentrations of dissolved organic carbon, suspended sediment, and certain metals. The increase in sediment loads can reduce reservoir storage capacity and adversely affect downstream aquatic ecosystems. Empirical observations show a wide range of watershed response to wildfires (e.g. Delp, 1968; Morris and Moses, 1987; Moody and Martin 2001; Benavides-Solorio, 2003), so there is an urgent need to quantify and predict: (1) the relative likelihood of high- or moderate-severity wildfires in different forest types across Colorado; (2) the changes in site productivity, runoff, and water quality following a wildfire; (3) the impact of repeated forest thinning on site productivity, runoff and water quality as compared to wildfires; and (4) the effectiveness of different burned-area emergency rehabilitation treatments.

With respect to the first need, different forest types and stand conditions have different probabilities for burning at high-severity, but the probability of large, high-severity wildfires are reasonably well known for only a few locations and forest types. For example, high or moderate severity fires are generally much more frequent in ponderosa pine than spruce-fir forests, but there is considerable variability in the recurrence intervals of high-severity fires within the different ponderosa pine sub-types (Romme et al., 2002). Since fire severity is a critical control on post-fire runoff and erosion rates (Benavides-Solorio, 2003), there is an urgent need to determine the risk of high-severity fires for the major forest types in Colorado as a function of stand density and location. This will probably require a combination of fire behavior modeling and dendrochronology studies.

The second need is for a better understanding of post-fire runoff and erosion rates for the forest types of primary concern. Recent studies on the Buffalo Creek fire (e.g., Moody and Martin, 2001), the Bobcat fire (Benavides-Solorio, 2003; Kunze, 2003), and other fires have greatly increased our knowledge of post-fire erosion rates in the ponderosa pine zone in the Colorado Front Range. The problem is that we have almost no data on post-fire runoff and erosion rates in other forest types, or for ponderosa pine forests outside of the Front Range. We also have relatively few water quality data taken immediately below burned areas as opposed to water supply intakes at downstream locations. A detailed understanding of the effects of wildfires on water quality will require intensive sampling within or immediately below the burned area. In addition to the standard constituents such as turbidity and total suspended solids, there is a need to understand the effects of wild and prescribed fires on dissolved organic carbon and metals such as manganese, as these constituents can affect human health and public acceptability.

The collection of post-fire erosion and water quality data is logistically difficult for several reasons. First, we cannot know when and where a wildfire is likely to occur, so it generally is not possible to have good pre-fire data. Second, the monitoring should begin immediately after the fire, as this is when the largest changes in water quality are likely to occur. The problem is that it is very difficult to have both funding and trained personnel available immediately after a fire. Third, there often is only limited or poor access to a burned area, or the burned area may be closed until the fire is completely controlled. Fourth, the burned area may be spread across portions of several watersheds rather than covering most of one or more watersheds. As in the case of timber harvest, a change in water quality is much easier to detect and quantify when the majority of a watershed has been "treated". The net result is a lack of data on the immediate post-fire response, even though this is the period of greatest concern to resource managers and the public. A concerted effort will be needed to establish monitoring programs that can be implemented immediately after wildfires.

A series of post-fire monitoring programs need to be established in order to quantify the changes in runoff, erosion, and water quality after wildfires in different vegetation zones. The goal is to develop and calibrate models that can be used to predict the likely changes in runoff, erosion, and water quality after wildfires. In order to apply such models to different sites, the measured changes in runoff, erosion, and water quality will have to be related back to the site characteristics such as slope, fire severity, rainfall erosivity, soil texture, and vegetation type. The resulting models could be used to estimate the magnitude of change that might be expected from different sites should a wildfire occur. By combining these models with spatially-explicit data on the probability of a high-severity wildfires and information on the value of downstream resources, one could predict which areas are at greatest risk. These assessments are needed to ensure that fuel reduction efforts are directed to the areas at greatest risk.

A third research need is to quantify the effects of different fuels treatments on runoff, erosion, water quality, and future wildfire risk. There appears to be an implicit presumption that mechanical thinning and prescribed fires have minimal adverse effects with respect to runoff, erosion, site productivity, and water quality, but there are very few studies to substantiate this. Hence there is an urgent need to document the effects of different thinning and fuels treatments on erosion rates and water quality in different forest types. We also need to determine the reduction in wildfire risk from a given treatment. For example, substantial thinning programs had been undertaken in the area around Cheesman Reservoir, but these areas still burned in the Hayman fire (Romme et al., 2002). In other portions of the Hayman fire, prescribed burning and forest thinning did appear to reduce fire severity and the rate of spread (Romme et al., 2002).

A full evaluation of the costs and benefits of fuels reduction programs will require longer-term comparisons between the effects of wildfires versus repeated fuels treatments. In other words, the presumption is that programs to reduce fuels cause little or no increases in runoff and erosion, so these programs have an overall net benefit as compared to the potential adverse effects after wildfires. However, the fuel reduction treatments may need to be repeated on a regular basis, so one cannot directly compare the runoff and erosion rates from wildfires to the rates from a single treatment to reduce wildfire risk. Instead, one has to integrate the effects of all the fuel treatments that would occur over an extended time period, and compare this to the predicted changes in runoff and erosion from the wildfires that should occur over the same time period in the absence of any fuel reduction treatments. Both portions of this assessment require a determination of the probabilities of the different severity wildfires that could occur with and without repeated fuel reduction treatments. The final step in the evaluation is to calculate the probability-weighted long-term differences in runoff, erosion, and water quality for each scenario (fuel treatments vs. wildfire), and then add in the economic costs of the fuel reduction program and probability-weighted costs of the different wildfire scenarios. Other than a recent study for California chaparral (Wohlgemuth, 2000), such longer-term evaluations of the overall benefits of fuel reduction programs have not been done.

A final research need is to evaluate the effectiveness of different post-fire emergency rehabilitation treatments. The effectiveness of such treatments was stated to be largely unknown in a recent USDA Forest Service publication (Robichaud et al., 2000), but this situation is beginning to change. A just-completed study on the Bobcat fire southwest of Fort Collins showed that mulching was the most effective treatment to reduce post-fire erosion rates. The aerial and hand seeding treatments had no detectable effect on the amount of vegetative cover or post-fire erosion rates. Contour felling was found to reduce erosion rates from smaller storms if the sediment storage capacity had not already been filled. The effectiveness of contour felling was limited by the poor installation in most of the study sites (Wagenbrenner, 2003).

After the Hayman fire the USDA Forest Service has spent approximately $17,000,000 on rehabilitation treatments, and the Denver Water Board spent approximately $5,000,000. Given these levels of expenditure, there clearly is a need to determine the effectiveness of different rehabilitation techniques under different site conditions and storm intensities.

6.3. Effects of Roads on Water Quality

Chapter 3 identified roads as the primary source of sediment in most forested areas. Road sediment production is often correlated with the amount and type of traffic as well as the frequency of grading (e.g., Reid and Dunne, 1984; Luce and Black, 2001). An increase in the rates of forest harvest or fuel treatments is likely to increase the amount of traffic on roads and the frequency of grading. In some cases it may be necessary to reopen closed roads or build new temporary roads. The road erosion literature suggests that each of these actions will increase erosion rates, but there are very few road erosion studies from Colorado. Empirical observations and studies from other areas indicate a wide range of road erosion rates, depending factors such as the road design, amount and type of traffic, and number and type of storm events. It follows that there is a need to evaluate road erosion rates in areas proposed for timber harvest or fuel treatments. It would also be helpful to collect similar data from the extensive unpaved road networks in the rural-urban interface, as these roads may have much higher erosion rates due to higher traffic loads and more frequent grading. The resulting field data can be used to test which – if any – of the existing road erosion models might be most applicable to Colorado.

The second, and possibly more important research need, is to develop tools to predict the delivery of sediment from unpaved roads into and through the stream channel network. It can be argued that erosion from unpaved roads is only an important concern when that sediment is moving off-site. The problem is that there are very few studies that have examined sediment delivery from roads to streams (e.g., Wemple et al., 1996; Montgomery, 1996). We know of only one study in Colorado that explicitly studied the delivery of sediment from roads to the stream network (Campbell, 1984). Sediment yields on Fool Creek were measured for 1-2 years after the roads were constructed but prior to timber harvest, but it is not clear whether the low sediment yields were due to a low road erosion rate or low delivery into and through the stream network. Data on road connectedness and the delivery of sediment to the stream network are needed to develop predictive models and management guidelines.

Until we have data to indicate otherwise, forest management plans and development proposals need to explicitly consider erosion from unpaved roads and the delivery of this sediment to the stream network.

Chapter 7
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