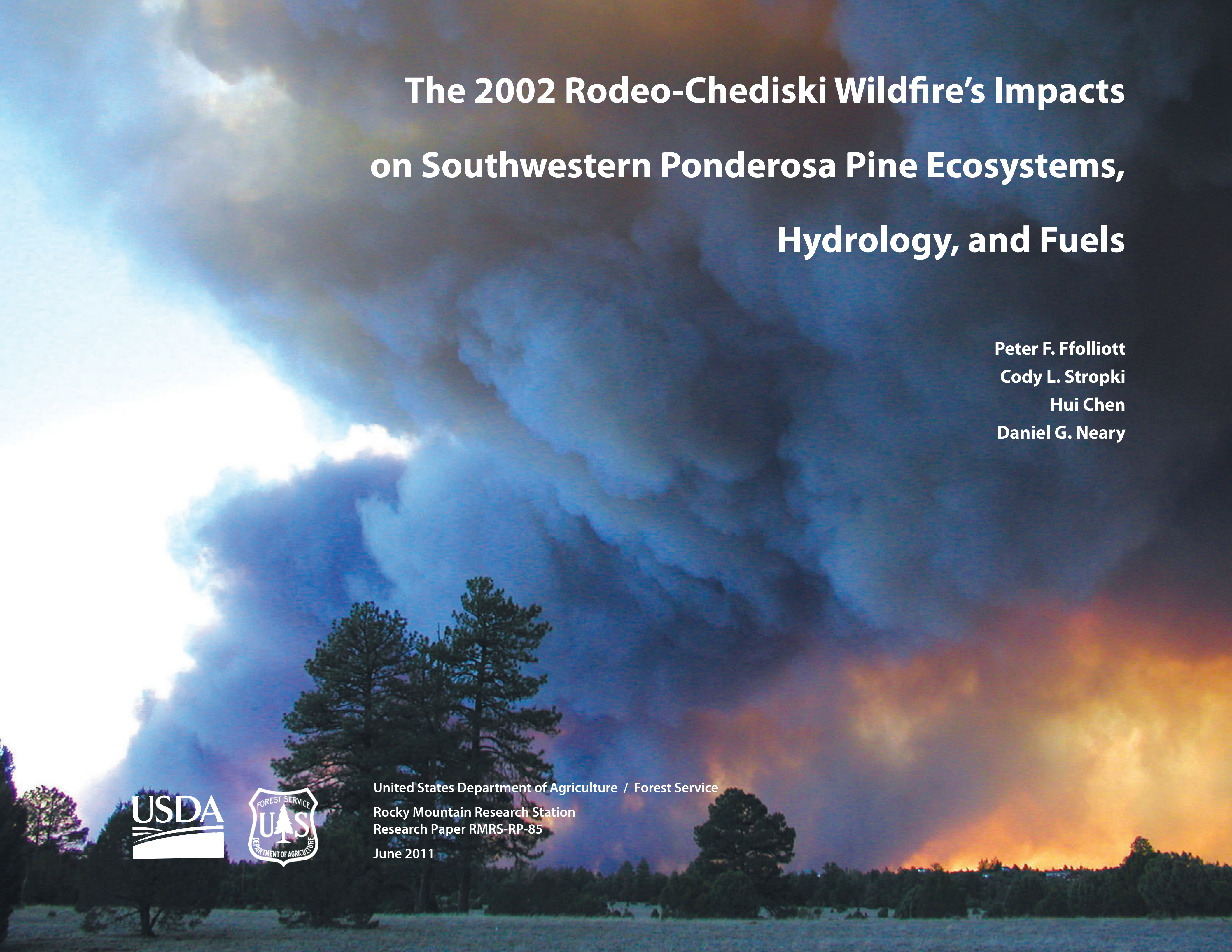

The 2002 Rodeo-Chediski Wildfire's Impacts on Southwestern Ponderosa Pine Ecosystems, Hydrology, and Fuels

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Abstract

The Rodeo-Chediski Wildfire burned nearly 462,600 acres in north-central Arizona in the summer of 2002. The wildfire damaged or destroyed ecosystem resources and disrupted the hydrologic functioning within the impacted ponderosa pine (*Pinus ponderosa*) forests in a largely mosaic pattern. Impacts of the wildfire on ecosystem resources, factors important to hydrologic functioning, peak stormflow events and water quality constituents, and loadings of flammable fuels were evaluated on two watersheds in a ponderosa pine forest that was exposed to the burn—one experienced a high severity (stand-replacing) fire (Watershed A), and the other was exposed to only a low severity (stand-modifying) fire (Watershed B). Cumulative impacts of the wildfire on ecosystem resources, hydrologic functioning, and flammable fuels were more pronounced on Watershed A. Recovery of the Stermer Ridge watersheds from the Rodeo-Chediski Wildfire has been related to the respective fire severities that the two watersheds experienced. Watershed A converted from ponderosa pine to grasses, forbs, and a few shrubs. Recovery of the hydrologic functioning on this watershed has begun on a limited scale, but it is anticipated that the overall hydrologic functioning of Watershed A will not approach pre-fire conditions for many years. Flammable fuels represented by standing trees have been eliminated on Watershed A, but there has been an increase in stem sections, branches, twigs, and herbaceous fuels on the forest floor. While the possibility of a future crown fire has declined, the potential for surface fire remains. Much of Watershed B is slowly recovering from the impacts of the wildfire. Much of the hydrologic functioning of this watershed is also returning slowly to its pre-fire level. The post-fire loadings of flammable fuels were largely unchanged from their pre-fire estimates. Watershed B remains vulnerable to future wildfire events as a consequence.

Keywords: flammable fuels, fire severity, hydrologic functioning, ponderosa pine, Rodeo-Chediski Wildfire

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Figure 1—The Rodeo-Chediski Wildfire evolved from two separate human ignitions on the White Mountains-Fort Apache Reservation in northeastern Arizona that merged into one inferno. (*Photo courtesy of the Apache-Sitgreaves National Forest.*)

Introduction

The 462,600-acre Rodeo-Chediski Wildfire of 2002 was the largest known in Arizona's history, and it was the ninth largest wildfire in the United States in terms of the acreage impacted (National Interagency Fire Center 2009). It damaged or destroyed ecosystem resources, disrupted hydrologic functioning, and altered the loadings of flammable fuels on much of the ponderosa pine forest that was exposed to the burn. Once a source of trees for processing into wood products, these forests are now vital in providing watershed protection in a water-deficient region, forage for domestic livestock and indigenous herbivores, habitats for mammals and avifauna, and sites for outdoor recreational activities.

An opportunity to study the impacts of this wildfire on ecosystem resources, hydrologic functioning, and flammable fuels presented itself on two first-order watersheds in a ponderosa pine forest. Watershed A was exposed to a high severity (stand-replacing) fire, and Watershed B was exposed to a low severity (stand-modifying) fire. The extensive area that was burned precluded establishment of a suitable unburned (control) area in the vicinity of the two watersheds for comparison purposes. This five-year study (2002 through 2007) was initiated shortly after cessation of the Rodeo-Chediski Wildfire and continued through the implementation of salvage cutting and fuel reduction treatments on Watershed A. Impacts of the wildfire were evaluated on:

- Ecosystem resources including tree overstories; herbaceous understories and shrub species; and wildlife species and their habitats.
- Factors of importance to hydrologic functioning such as the formation of water-repellent soils; soil movement, including both soil erosion and soil deposition; and peak stormflow events and water quality constituents.
- Loadings of flammable fuels, including the fractions of standing trees, downed woody materials, litter and duff accumulations, and herbaceous plants.

The results of this study complement the findings from earlier studies on the impacts of wildfire on ecological resources, hydrologic functioning, and flammable fuels in southwestern ponderosa pine forests (Baker 1990; Campbell and others 1977; Harrington 1987; Ice and others 2004; McHugh and Kolb 2003; Neary and others 2005; Pearson and others 1972). This report contains brief summaries of some of those studies. The information in this report should help managers evaluate the impacts of large-scale wildfires on ecosystem resources, hydrologic functioning, and flammable fuels and then plan effective post-fire rehabilitation activities.

Rodeo-Chediski Wildfire

The Rodeo-Chediski Wildfire evolved from two separate human-induced ignitions on the White Mountains-Fort Apache Reservation in northeastern Arizona that merged into one inferno (fig. 1). Arson was the cause of the Rodeo fire that started on June 18, 2002, a few miles from Cibecue, a small village on the Reservation. The Chediski fire was set as a signal by a lost hiker a few days later. The second fire quickly spread out of control and eventually merged with the still out of control Rodeo wildfire on June 23rd. The re-named Rodeo-Chediski Wildfire then burned onto the adjacent Apache-Sitgreaves and Tonto National Forests and through many small communities before it was contained on July 7th at a suppression (fire fighting) cost of about $45 million (Morton and others 2003; Neary and Zieroth 2007). Nearly 500 homes and other buildings were destroyed and over 30,000 people were forced to flee. Property losses, losses of ecosystem and cultural resources, and rehabilitation efforts increased the monetary costs of the wildfire to nearly $175 million.

Rehabilitation efforts began shortly after the wildfire was extinguished. Detention dams and other diversions were constructed to re-route (away from infrastructures) overland flows of water that originated from the post-fire summer rainstorms. Culverts were removed to reduce road damage during the anticipated flooding events. Helicopters ferried bales of straw to areas that were susceptible to increased soil erosion rates. Straw was spread onto the ground to reduce the impacts of the post-fire summer rains on the bare soil surface. Aerial seeding of grasses and other herbaceous plants on the erosion-prone sites proved ineffective. Most of the seed washed downslope into stream corridors in the initial post-fire rainstorms and few germinated on the slopes. Fire-damaged but salvageable trees 12 inches in diameter at breast height (dbh) and larger with merchantable value, but not expected to survive, were harvested on the White Mountains-Fort Apache Reservation beginning in the late fall of 2002. Salvage cutting and fuel reduction treatments were initiated on the Apache-Sitgreaves and Tonto National Forests in the summer of 2005. Trees smaller than those harvested in the salvage cutting were cut in the fuel reduction treatments.

Stermer Ridge Watersheds

Two mostly homogeneous watersheds, 60 acres each, that are located along Stermer Ridge at the headwaters of the Little Colorado River were the sites chosen for this study (fig. 2). These watersheds, designated A (high severity wildfire) and B (low severity wildfire), are similar vegetatively, physiographically, and hydrologically. The watersheds were originally instrumented in 1972/1973 by the School of Natural Resources and the Environment, University of Arizona, and the Rocky Mountain Research Station, U.S. Forest Service, to obtain ecological and hydrologic information on ponderosa pine forests found on soils derived from sedimentary parent materials (Ffolliott and Baker 1977). The information obtained on these watersheds

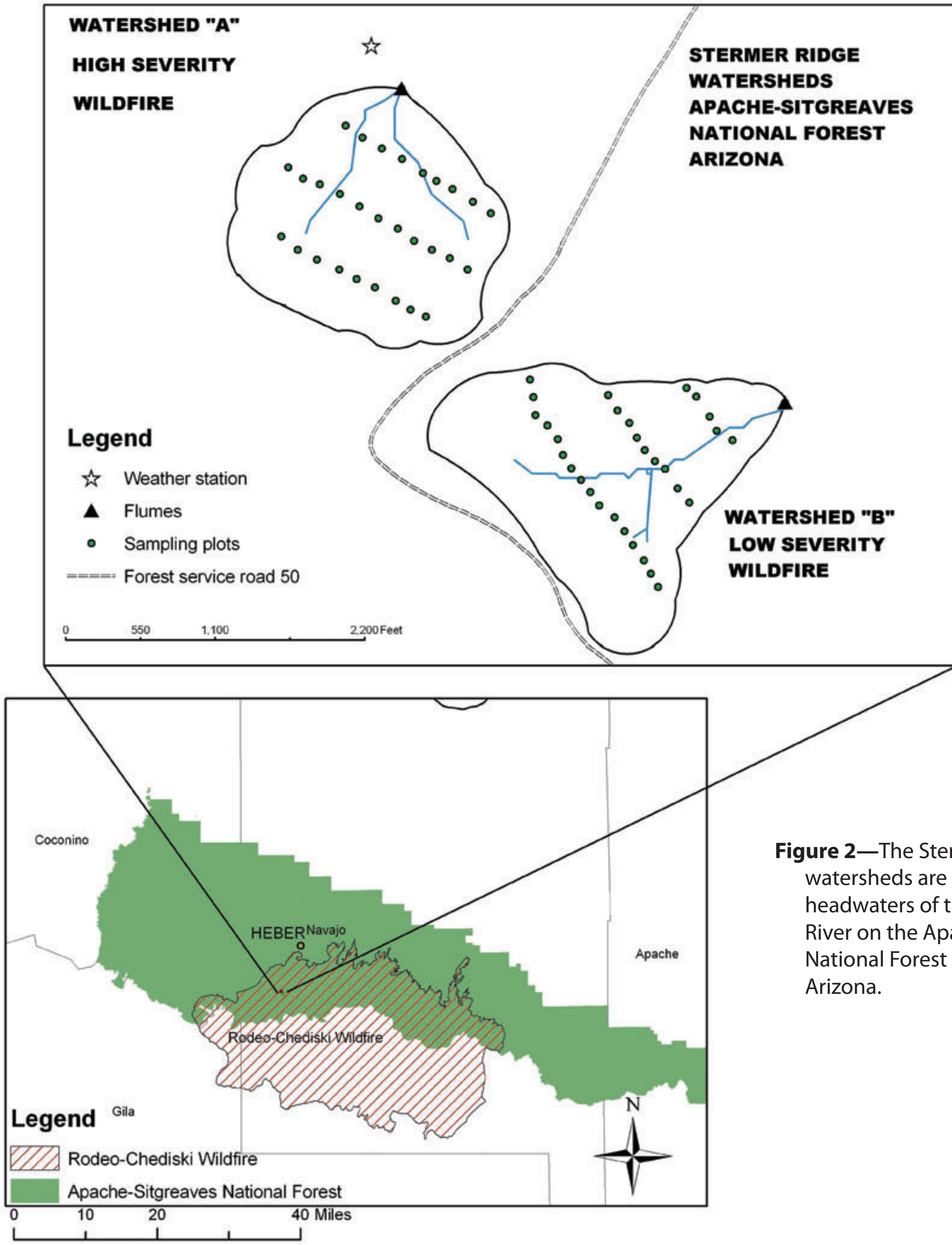


Figure 2—The Stermer Ridge watersheds are located at the headwaters of the Little Colorado River on the Apache-Sitgreaves National Forest near Heber, Arizona.

complemented the watershed-based research program on basaltic soils in the Southwestern Region (Baker 1999; Fox and others 2000).

Pre-fire overstories of ponderosa pine trees on the two watersheds had been selectively harvested for merchantable timber in several earlier cuttings. The most recent timber harvesting operation before the Rodeo-Chediski Wildfire removed about 45 percent of the merchantable trees by group selection in the early 1960s. A few intermingling Gambel oak (*Quercus gambelii*) and alligator juniper (*Juniperus deppeana*) trees were also present on the watersheds. Graminoids, forbs, and shrubs found in southwestern ponderosa pine forests (Clary 1975; Ffolliott and Baker 1977; Ffolliott and Gottfried 2008, and others) comprised pre-fire herbaceous understories. Squirreltail (*Elymus elymoides*), muttongrass (*Poa fendleriana*), Arizona fescue (*Festuca arizonica*), blue grama (*Bouteloua gracilis*), prairie Junegrass (*Koeleria macrantha*), mountain muhly (*Muhlenbergia montana*), black dropseed (*Sporobolus interruptus*), and sedge (*Carex* spp.) were the main grass and grasslike plants on the watersheds. Among the common forbs were showy goldeneye (*Heliomeris multiflora*), Cuman rag-weed (*Ambrosia psilostachya*), Wright's deervetch (*Lotus wrightii*), Cooley's bundleflower (*Desmanthus cooleyi*), yellow sweetclover (*Melilotus officinalis*), trailing fleabane (*Erigeron flagellaris*), and western yarrow (*Achillea millefolium* var. *occidentalis*). Broom snakeweed (*Gutierrezia sarothrae*), the shrub-form of Gambel oak; New Mexican locust (*Robinia neomexicana*); Mexican cliffrose (*Purshia mexicana*); and Fendler's ceanothus (*Ceanothus fendleri*) were among the scattered half-shrubs and shrubs. The Stermer Ridge watersheds provided food resources and protective cover for a variety of wildlife species, including indigenous ungulates, small mammals, and birds.

Watersheds A and B are characterized by flat topographies with few slopes that exceed 10 percent. Elevations range from 6800 to 7000 ft. Soils derived from cretaceous materials lying beneath the watersheds are classified in the McVickers series with sandy loam surface textures. According to Hendricks (1985), the soils in this series are moderately deep to deep and are well drained. They have a high available water capacity, a slow permeability, and a high shrink-swell potential; and are susceptible to slight to moderate erosion rates. The watersheds normally receive in excess of 20 inches of precipitation annually. Nearly 65 percent of this precipitation occurs from October to April, much of it as snow or low-intensity, long-duration winter rains (fig. 3). The remaining precipitation falls in mostly high-intensity, short-duration summer rainstorms from early July through the middle of September. However, a drought was impacting the watersheds and, more generally, the Southwestern Region before the onset of the Rodeo-Chediski Wildfire and continued to prevail throughout the study period. Annual precipitation on the Stermer Ridge watersheds during this drought was less than 80 percent of the normally expected precipitation. Nearly 75 percent of the volume of the pre-fire intermittent stormflows from the Stermer Ridge watersheds occurred as a result of snowmelt-runoff events or winter rains, while the summer rains generated the highest peak stormflows.

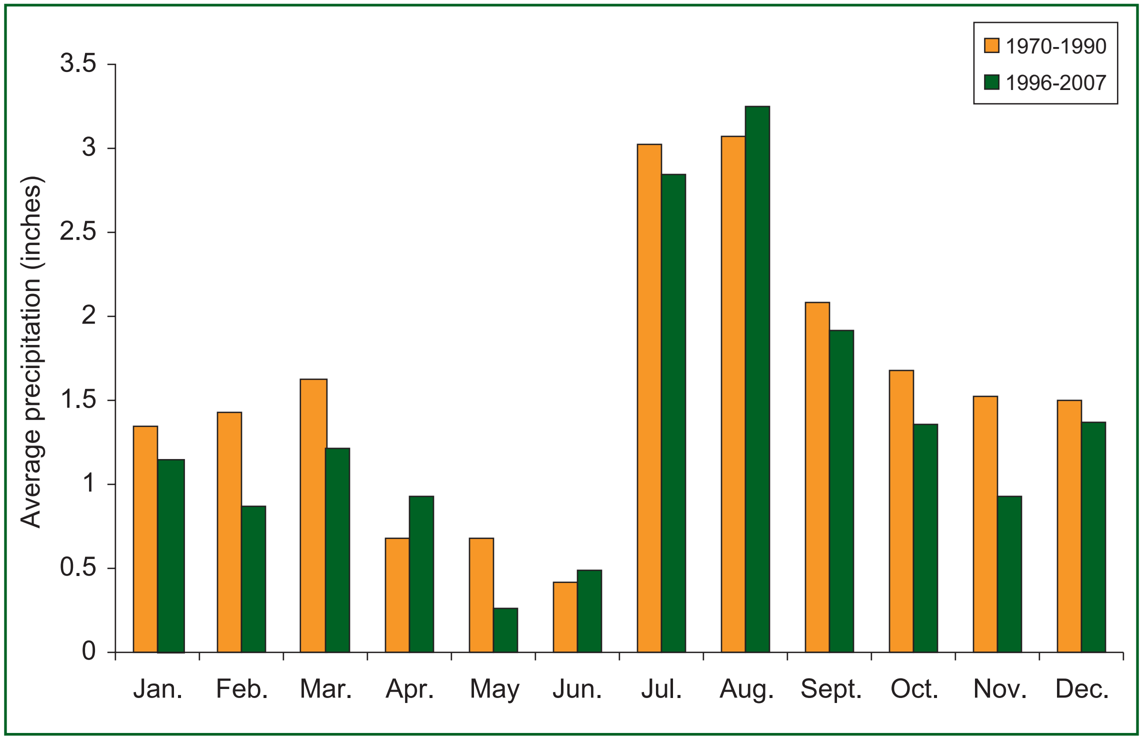


Figure 3—Average monthly precipitation amounts in the vicinity of the Stermer Ridge watersheds before (1970 to 1990) and during (1996 to 2007) the drought conditions that prevailed in the study. The data compiled for this figure were obtained from the Black Mesa Ranger Station, U.S. Forest Service—the nearest long-term weather station (approximately 7 miles) to the watersheds.

A classification system that relates fire severity to the soil-resource response of burning (DeBano and others 1998; Hungerford 1996; Ice and others 2004) was the basis for assessing fire severity at the primary sampling plots on the watersheds (see the *Study Protocols* section) shortly after cessation of the wildfire. These classifications were extrapolated to a watershed basis (Neary and others 2005; Ryan and Noste 1985; Wells and others 1979) to determine the portions of the watersheds that were unburned or had burned with a low, moderate, or high severity fire. This extrapolation procedure indicated that Watershed A had experienced a high severity (stand-replacing) fire, while Watershed B was exposed to only a low severity (stand-modifying) fire. A road between the two watersheds that was maintained by the U.S. Forest Service (FS-50) acted as a partial break by containing the high severity burn to only one of the watersheds.

Study Protocols

The Stermer Ridge watersheds were "moth-balled" in late 1977 after completion of the earlier studies (Ffolliott and Baker 1977). However, the control sections (3-ft H-flumes) on both watersheds that were used for monitoring stormflow events were left in place in anticipation of future needs. Following the Rodeo-Chediski Wildfire, the control sections were refurbished and re-instrumented with water-level recorders, and a weather station was re-established. A set of 30 sample plots were relocated that were originally installed on each of the watersheds in a systematic sampling scheme with multiple random starts (Shiue 1960). These plots were used to obtain ecological and hydrologic information in the earlier studies (Ffolliott and Baker 1977). The sampling scheme combined random and systematic elements and, as such, allowed for computing measures of variation while ensuring that a representative sample was made of the area. The plots provided the primary sampling framework in evaluating the impacts of the wildfire on the ecosystem resources, hydrologic functioning, and flammable fuels of the watersheds.

Ecosystem Resources

Tree Overstories

It was necessary to reconstruct the tree overstory structures on the Stermer Ridge watersheds before the Rodeo-Chediski Wildfire to form a basis for evaluating impacts of the fire on the overstories. Two approaches were taken to obtain these reconstructions. One approach consisted of tallying trees (both alive and killed outright by the fire) that were 6 inches and larger in dbh at the sample plots by point sampling techniques (Avery and Burkhart 2002) with a basal area factor of 25. The tallies included burned-out root cavities of trees consumed by the wildfire (fig. 4). The dbh of pre-fire trees that occupied those cavities was estimated from previously established relationships between dbh and stump diameters (Myers 1963). Diameters of the cavities were assumed to approximate the stump diameters of the pre-fire trees.

Figure 4—Burned-out root cavities of trees consumed by the Rodeo-Chediski Wildfire on the Stermer Ridge watershed that was burned by a high severity fire. (*Photo by Peter F. Ffolliott*).

The second approach involved sequential interpretations of a series of computer-simulated stand-table projections from 1977, when the watersheds were retired from active use, to before the wildlfire occurred in 2002. The STAND simulator used in this effort (Ffolliott and others 1988) was a simplified version of the Timber Resource Analysis System (TRAS), a widely used computer program for estimating forest resources in the United States (Alig and others 1982; Larson and Goforth 1970).

Reconstructions obtained by point sampling techniques and reconstructions simulated by the simplified version of TRAS were similar in their tree distributions by dbh classes and had only insignificant discrepancies (Ffolliott and others 2008). Therefore, the average of the tree distributions obtained by the two reconstructions was selected to represent the structures of the pre-fire tree overstories. The reconstructed overstory structures were summarized in terms of the number of trees per acre by 2-inch dbh classes (that is, the 6-inch dbh class included trees 5.0 to 6.9 inches, the 8-inch dbh class included trees 7.0 to 8.9 inches in dbh, and so forth), and were then grouped into two size classes for analysis purposes. The size classes coincided with the categories specified by Smith and others (2004) for poles (at least 6 inches in dbh but smaller than sawtimber) and sawtimber (12 inches in dbh and larger). Sawtimber trees were further classified into small sawtimber (up to 18 inches in dbh) and large sawtimber (20 inches in dbh and larger). Small and large sawtimber trees are often tallied separately in timber inventories and in other mensurational studies in the Southwestern Region because of their different forms.

Trees that initially survived the wildfire were classified by crown damage in four categories:

- No crown damage.
- Less than one-third of the crown scorched.
- Between one-third and two-thirds of the crown scorched.
- More than two-thirds of the crown scorched.

Similar criteria have been used in classifying fire damage and to predict post-fire mortality of southwestern ponderosa pine trees by Dieterich (1979), Herman (1954), and McHugh and Kolb (2003), and in other studies reviewed by Fowler and Sieg (2004). Annual observations of the subsequent mortality of the fire-damaged trees were made from 2002 through 2006. However, the estimates of mortality in 2006 were confounded because of the salvage cutting and fuel reduction treatments that were initiated on the watershed that was burned by a high severity fire.

Stocking of Ponderosa Pine Seedlings

Ponderosa pine seedlings (trees less than 1inch in dbh) were tallied on 30 mil-acre plots superimposed on the primary sample plots to determine the impact of the wildfire on stocking of this reproduction. A plot was classified as stocked if (at least) one seedling was tallied. Otherwise, the plot was considered as not stocked. The stocking of ponderosa pine seedlings was recorded shortly after the wildfire was extinguished and then again in 2004 and 2006, two and four years after the burn, to assess the longer-term post-fire stocking conditions on the watersheds.

Species Compositions and Production of Herbaceous Plants and Shrubs

Species were noted, and production (above-ground biomass) of the herbaceous plants (grasses and forbs combined), grass and forb components (considered individually) of the herbage, and shrubs in the understories of the Stermer Ridge watersheds was estimated on 9.6-ft^2 plots centered over the primary sample plots. Estimates of production were obtained by applying the double-sampling method outlined originally by Pechanec and Pickford (1937) and later by Cook and Stubbendieck (1986), Holechek and others (2004), and Shoop and McIlvain (1963). Field estimates of green weights of the plants were made in late spring and early fall to reflect the production of early-growing (cool-season) and late-growing (warm-season) plants, respectively. Growth of early-growing plants is responsive largely to winter precipitation and the consequent soil moisture conditions, while the growth of late-growing plants is related mostly to summer rainstorms (McPherson 1992, 1997). Estimates of herbage production were obtained initially in the fall of 2002, after the wildfire had been extinguished, and continued through the spring of 2007. Samples of the plants tallied were collected on 15 temporary 9.6-ft^2 plots located adjacent to a sub-sample of the primary sample plots at the time that production estimates were made to develop the correction factors necessary to convert the field estimates of green weights to actual (oven-dried) weights.

Frequency distributions of the estimated production of the herbaceous plants and grass and forb components of the herbage were mostly non-normal according to Shapiro-Wilk tests of normality (Yazici and Yolacan 2007). Transformations of the data sets failed to normalize the non-normal frequency distributions. Occurrences of statistical differences in the production of the plant groups sampled, therefore, were evaluated by distribution-free non-parametric tests. Applications of these non-parametric tests are independent of the structure of the frequency distributions involved. The non-parametric tests applied in this study are analogues to corresponding parametric methods of data analyses (table 1). Details of the tests and their interpretations are outlined in Zar (1999) and other references on non-parametric methods of statistical analysis.

Separate analyses were conducted to determine whether there were statistical differences in the post-fire production of the herbaceous plants combined and grass and forb components of herbage on the Stermer Ridge watersheds. These analyses were conducted by applying Mann-Witney tests for each season that the production estimates were obtained. This test was deemed appropriate because the representative data sets obtained on the two watersheds were considered to be independent of each other. Differences between the production of early-growing and late-growing herbaceous plants, grasses, and forbs on each of the watersheds were evaluated by using the Wilcoxon-paired-sample test in each year of the study that the production of both the early-growing and late-growing plant groups were estimated on the watersheds (in 2003, 2004, 2005, and 2006). This test was considered appropriate for this purpose because the production of early-growing plants was not totally independent of the production of late-growing plants. Some of the plant species that were sampled initiated their growth in early spring and then continued to grow into late summer or early fall and, as a consequence, were included in both seasonal estimates of production. Nemenyi tests were used to determine whether statistical differences occurred in the production of early-growing plants for each season that the production of the plants was estimated. This test was also applied in analyzing differences in the production of late-growing plants when the production was estimated.

Table 1—Non-parametric tests used in this study and corresponding parametric methods of data analyses.

Sample	Non-parametric test	Parametric method
Two independent samples	Mann-Witney test	Two-sample t-test
Two dependent samples	Wilcoxon-paired-sample test	Paired-sample t-test
Multiple Comparison	Nemenyi test	Tukey test

All tests of statistical differences in the production of the plant groups were evaluated at a 0.10 significance level. However, because the grass and forb components of the herbage were nested within the overall tests of all herbaceous plants combined, the individual tests of those two components were evaluated separately at a 0.05 level of significance to maintain the overall 0.10 level of significance in accordance with a Bonferroni adjustment.

Differences in the production of shrub species on the Stermer Ridge watersheds were not analyzed analytically because of the limited frequencies of occurrence and low levels of production of the plants that only occurred in the latter stages of the study.

Utilization of Forage Plants

Utilization of forage plants, that is, the herbaceous plants and shrubs that provide food for grazing livestock and indigenous ungulates, was estimated ocularly (Cook and Stubbendieck 1986; Holechek and others 2004) on the 9.6-ft^2 plots at the time that the plant production was estimated. No differentiation was made among the kinds of herbivores that consumed the forage.

Indigenous Ungulates and Small Mammals

Use of the post-fire habitats on the Stermer Ridge watersheds by elk (*Cervus elaphus*) and mule deer (*Odocoileus hemionus*) was estimated by counting the accumulations of their fecal pellet groups in the 0.01-acre plots centered over the primary sample plots. Pellet group counts had also been used to estimate population numbers of these ungulates in earlier studies (Clary and Larson 1971; Eberhardt and White 1979; Larson and others 1986; Neff 1968, 1972; Rowland and others 1984; Smith 1968). However, problems in obtaining reliable estimates of population numbers using this technique have been pointed out by Braun (2005), Collins and Urness (1979), Davis (1982), Neff (1968), Rowland and others (1984), and others.

One reason that the counting pellet groups can lead to inaccuracies in estimating population numbers is that it is necessary to assume a constant defecation rate between the sexes and within age classes of the species in the population to be estimated. This assumption is not always valid. For example, seasonal diets of elk and mule deer often vary, with less moisture and digestible matter in winter foods than foods in summer, resulting in lower defecation rates in the winter. Furthermore, it is not always known whether a count of 10 pellet groups represents 10 elk or mule deer that defecated once or 1 elk or mule deer that defecated 10 times without validation information such as that obtained from a sub-sample of radio-marked animals. Because of these and other possible problems encountered with the use of this technique, the counts of elk and mule deer pellet groups obtained in this study were interpreted only as indices of presence (occurrence) of the ungulates on the watersheds.

Earlier depositions of elk and mule deer pellet groups were cleared from the 0.01-acre plots when the primary sample plots were re-located in the fall of 2002 because the time of their accumulation was unknown. Elk and mule deer pellet groups were then counted on, and cleared from, the plots beginning in the spring and fall of 2003 and continuing to the spring of 2007. Differentiating elk from mule deer pellet groups was not a problem in this study.

Counts of the elk and mule deer pellet groups obtained in this study fit negative binomial frequency distributions. This frequency distribution had been determined by other researchers to be appropriate in earlier studies involving elk and mule deer pellet group counts (Bowden and others 1969; McConnell and Smith 1970; Rowland and others 1984). Two parameters, m (the average frequency of pellet groups per plot) and k (a positive exponent that measures dispersion), characterize a negative binomial distribution. Procedures outlined by White and Bennetts (1996) and White and Eberhardt (1980) for analyzing the differences in count data fitting the negative binomial frequency distribution, such as the counts of elk and mule deer pellet groups in this study, were applied. It was assumed that the m and k values differed in the analyses because it was anticipated that the elk and mule deer use of the habitats on the two watersheds might also differ. Tests for statistical differences in the total counts of the pellet groups on each of the watersheds for a sampling period were evaluated at a 0.10 level of significance.

Occurrences of cottontail (*Sylvilagus auduboni*) on the watersheds following the wildfire were estimated by counting the individual fecal pellets deposited on the 0.01-acre plots that were used to count the elk and mule deer pellet groups. This technique had been used in earlier studies (Cochran and Stains 1961; Kundaeli and Reynolds 1972; and others) to obtain estimates of cottontail population numbers. However, the problems encountered in obtaining reliable estimates of population numbers for elk and mule deer from counts of fecal pellets also applied to the cottontail pellet counts. Similar to interpretations of the counts of elk and mule deer pellet groups, therefore, the cottontail pellet counts obtained in this study were also interpreted as indices of presence.

After earlier accumulations of individual pellets were cleared from the 0.01-acre plots when the primary sample plots were re-located, cottontail pellets were counted on and cleared from the plots from the spring and fall of 2003 through the spring of 2007. However, the few cottontail pellets that were eventually counted in the study precluded a statistical analysis of the data sets obtained.

Twigs of ponderosa pine trees clipped in feeding activities by Abert's squirrel (*Sciurus aberti*), the common tree squirrel in southwestern ponderosa pine forests, were noted when found beneath trees that survived the Rodeo-Chediski Wildfire (fig. 5) at the time that elk, mule deer, and cottontail pellets were counted. The presence of the squirrel after the wildfire and its post-fire selection of feed trees, a key habitat component (Brown 1984; Ffolliott and Patton 1978; States and others 1988), were then inputted from this information.

Figure 5—Twigs clipped in feeding by Abert's squirrel (*Sciurus aberti*) beneath trees that survived the Rodeo-Chediski Wildfire on the Stermer Ridge Watershed B. (*Photo by Peter F. Ffolliott*)

Birds

Birds sighted in five-minute observation-periods at each primary sample plot on the watersheds were tallied by species and numbers according to established procedures (Braun 2005; Davis 1982; Ralph and others 1995). These tallies began a few minutes after the observer arrived at a plot to minimize the effects of disturbances caused by the observer moving onto the plot. Fall sightings were initiated in 2002 and continued in 2003, 2004, and 2005. Spring tallies were made from 2003 through 2007. All of the bird observations were obtained between 0800 and 1130 on clear or partly cloudy days with minimal wind movement. Summaries of these tallies provided a snap-shot picture of the birds on the watersheds at the time of their observations.

Species richness, that is, the number of bird species sighted (MacArthur and MacArthur 1961), was determined for each of the observation periods. Knowledge of the species richness of birds was supplemented by obtaining a number that represented species diversity (Shannon and Weaver 1948). The number (H') was calculated by:

$$H' = -\sum_{i=1}^{s} p_i \ln (p_i)$$

The variable p_i represents the proportion of the i^{th} species in a population of birds composed of s species. Larger numbers represent higher species diversities. Evenness (E) of the species tallied, or how equally abundant were the species, was determined by:

$$E = H'/\ln s$$

Larger values for species evenness were equated with higher diversity conditions. Values approaching 1 represented the highest level of species diversity.

Hydrologic Functioning

Water-Repellent Soils

Water-repellent (hydrophobic) soils impact the infiltration process of the hydrologic cycle in a similar manner as a hardpan layer might. Infiltration of water into the soil can be inhibited and often completely impeded, with much of the precipitation that reaches the soil surface accelerating the magnitudes and velocities of the resulting overland flows of water (Brooks and others 2003; DeBano and others 1998; Erickson and White 2008; Neary and others 2005). One mechanism that causes the formation of water- repellent soils is the distillation of organic compounds in the litter layer on the soil surface in the combustion phase of a fire (DeBano 1981, 2003; Doerr and others 2000; Neary and others 2005). It was assumed that this mechanism caused the formation of water-repellent soils on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire.

Occurrence of water-repellent soils was determined at each primary sample plot on the two watersheds using the water-drop penetration method outlined by Letey and others (2000). The initial measurements of water repellency were obtained in the fall of 2002, and continued until measurable water-repellent soils ceased to exist on each watershed. Measurements of water repellency were made in the spring and fall throughout the study to reflect occurrences and levels of the repellency following the periods of snowmelt runoff and winter rains (combined) and summer rainstorms, respectively, and were then summarized accordingly. Litter and duff accumulations were cleared to expose mineral soil for each measurement. A drop of distilled water was placed on the soil surface, and the time for the water drop to penetrate the soil surface was recorded (fig. 6). This procedure was then repeated because the precise location of a water- repellent layer at the plot (if a layer existed) was not known. The longest duration of the two times measured for the water drop to penetrate into the soil was related to the following criteria of the National Wildfire Coordinating Group (Clark 2001) to determine the level of water repellency:

- No repellency.
- Slight repellency, less than 10 seconds.
- Moderate repellency, 10 to 40 seconds.
- Strong repellency, more than 40 seconds.

Figure 6—Time for a drop of water (arrow) placed on the soil surface to penetrate into the soil was a measure of water-repellent (hydrophobic) soils at a sample plot on Watershed A. (*Photo by Cody L. Stropki*)

Transpiration and Infiltration

Transpiration was another component of the water budget that was altered by the Rodeo-Chediski Wildfire. To estimate the magnitude of this component, measurements of instantaneous transpiration rates were obtained by the sap flow method (Swanson 1994) shortly after the fire on three severely damaged and three undamaged ponderosa pine trees, 18 to 24 inches in dbh, on each of the watersheds.

The major impact of wildfire on water infiltration into the soil comes from the formation of water- repellent layers. Infiltration rates commonly decline from levels exceeding most peak rainfall intensities to zero. It was assumed that formation of water-repellent soils in the early years of the Stermer Ridge study inhibited the infiltration rates on the watersheds, with most of the precipitation reaching the soil surface contributing to surface runoff, thereby increasing the flows of overland water and movement of soil particles. Infiltration of rainfall into the soil was not measured directly in the study, and only by proxy as water repellency.

Soil Movement

The cumulative impacts of the loss of protective vegetation, formation of water-repellent soils, and increase in overland flows of water following a wildfire often result in increased soil erosion and soil deposition on the hillslopes of a burned watershed (Brooks and others 2003; Chang 2006; DeBano and others 1998; Neary and others 2005). Soil particles that are eroded from one site are either deposited downslope on another site or are transported into a stream channel to become sediment when water flows. The transport of soil particles and their eventual deposition within a watershed after a wildfire is linked closely to the matrix of severities within the fire perimeter and to the magnitudes, velocities, and timing of consequent overland flows of water. Though sediment accumulations in the stream channels were not measured in this study, measurements of post-fire soil erosion and soil deposition on the hillslopes of the watersheds were obtained.

Initial estimates of soil erosion were obtained on the Stermer Ridge watersheds shortly after the Rodeo-Chediski Wildfire occurred from the measurement of 23 soil pedestals on Watershed A and from 18 soil pedestals on Watershed B. Three capped pins were subsequently placed around each of the primary sample plots on the watersheds to obtain more precise measurements of soil erosion and deposition. Two pins were placed 3 ft upslope and one pin 3 ft downslope of a plot. Once these pins were installed, seasonal measurements of soil loss beneath the cap of the pins (soil erosion) or soil accumulation on top of the cap (soil deposition) were made in the spring and fall to characterize post-fire soil movements following periods of snowmelt runoff, winter rains (combined), and summer rainfalls, respectively. On occasion, there was no measurable change in the soil surface beneath the cap. It was assumed in these cases that either the magnitudes of soil erosion and soil deposition in the time interval between the measurements equaled each other or, what was less likely, neither soil erosion nor deposition occurred in the measurement interval.

The capped pins were re-set flush to the soil surface after each measurement to facilitate the subsequent measurements. Measurements obtained from the three pins surrounding a plot were averaged to estimate soil movement at the plot. A bulk density value that was developed from a set of soil samples collected on unburned sites near the Stermer Ridge watersheds was applied in converting the measurements of soil erosion and deposition to tons per acre on a watershed basis.

Measurements of soil erosion and deposition obtained in the study were analyzed separately because they represented separate processes of soil movement on a hillslope. Interpretations of the results of Shapiro-Wilk tests of normality indicated that the frequency distributions for both soil erosion and deposition were mostly non-normal in structure. Transformations of the data sets did not normalize these distributions. As a consequence, two of the non-parametric tests used in analyzing the production of herbaceous plants (Mann-Whitney and Nemenyi) were also applied in analyzing the measurements of soil erosion and deposition in this study. A minimum number of 12 plots with measurable soil erosion and a minimum of 4 plots with measurable soil deposition in each of the measurement periods were selected arbitrarily as a requirement to conduct statistical analyses of the respective processes. Plots with no measurable change in soil movement were excluded from the analyses.

Occurrences of statistical differences in the measurements of soil erosion rates and soil deposition between the two Stermer Ridge watersheds were determined by the Mann-Whitney test for each season in the study that the minimum numbers of measurements of these processes were available. This test was considered appropriate because estimates of the respective processes on the two watersheds were independent of each other. Differences between the measurements of soil erosion in the spring and in the fall and differences between the measurements of soil deposition in the spring and in the fall, on each of the watersheds were also analyzed by the Mann-Witney test for each year that the required number of measurements had been made. Because the respective measurements were considered independent of each other, the Mann-Whitney test was deemed suitable for the evaluations. Differences in the soil erosion measurements obtained in the spring and differences in the soil erosion measurements obtained in the fall on each watershed were evaluated (separately) by the Nemenyi test when the required measurements were available. The respective differences in seasonal measurements of soil deposition were also analyzed by the Nemenyi test. All of the tests of differences in soil movement were evaluated at an $\alpha = 0.10$ level of significance.

Peak Stormflows

Initial estimates of peak stormflows (ft^3/sec) following the Rodeo-Chediski Wildfire were obtained from observed high-water marks in or above the H-flumes on the watersheds following the rainfall events that occurred before the control sections could be re-instrumented with water-level recorders. Following the re-instrumentation, the depths of stormflows that passed through the flumes were obtained on the water-stage recorders. All estimated and recorded depth measurements were converted to rates of peak stormflow through interpretations of the established rating curve for the flumes. Only measurements of peak stormflows from early April through late November were obtained in this study because of the limited accessibility to the Stermer Ridge watersheds in the winter months and because of a lack of electronic recorders. Loss of the winter stormflows was not considered to be critical to the study since these flows had been generally low in the past in the Southwest unless a rare rain-on-snow event occurred. The highest peak stormflows originating on ponderosa pine watersheds of the Mogollon Rim are normally generated from high-intensity, short-duration summer rainfall events (Ffolliott and Baker 1977).

Water Quality Characteristics

Concentrations of sediment particles and dissolved nutrients are among the water quality characteristics of primary interest to hydrologists and watershed managers in the Southwestern Region. Unfortunately, it was not possible in this study to collect samples of stormflow water to measure those concentrations. The intermittent post-fire stormflows that originated on the watersheds generally occurred when people were not on-site, and collection basins for obtaining stormflow samples in the absence of people were not part of the instrumentation on the watersheds. However, samples of water flowing off of the larger area burned by the Rodeo-Chediski Wildfire were obtained intermittently by hydrologists and watershed managers further downstream to provide a general analysis of selected water quality constituents.

Loadings of Flammable Fuels

Knowledge of the impacts of the Rodeo-Chediski Wildfire on flammable fuels is helpful to managers in anticipating the frequency of occurrences, severities, and impacts of future wildfire events in southwestern ponderosa pine forests. However, the loadings of flammable fuels on the Stermer Ridge watersheds at the time of the wildfire were unknown. Therefore, estimates of those loadings were obtained from the information available from earlier studies on the watersheds (Ffolliott and Baker 1977; Ffolliott and others 1971). Loadings reported earlier in the literature provided an additional source of data (Harrington 1986; Sackett 1979; Sackett and Haase 1996). New measurements made in this study made up the third source of fuel loadings data. Estimates of the pre-fire fractions of standing trees, downed woody materials, litter and duff accumulations, and herbaceous plants and shrubs were compared to post-fire estimates of those fractions to evaluate the impacts of the Rodeo-Chediski Wildfire.

Standing Trees

Cubic-foot volumes of standing trees were tallied on the watersheds (Ffolliott and others 2008) and were converted to oven-dry weights by multiplying the volumes by the appropriate species-specific wood density values (Barger and Ffolliott 1964, 1965, 1971) to estimate the loadings of this fraction. Pre-fire approximations of the fraction were based on the reconstructions of tree overstories structures before the wildfire (see the *Tree Overstories* section). Post-fire estimates of the fraction obtained one and three years after the fire were based on knowledge of the impacts of the Rodeo-Chediski Wildfire on the reconstructions. Loadings of this fraction were separated into live trees and dead trees because the higher moisture contents of living trees increase their ignition time and decrease their rates of burning relative to dead trees (DeBano and others 1998; Pyne and others 1996). Moist fuels either do not burn or burn slowly and at lower temperatures than dry fuels, while dry fuels tend to burn hot, more completely, and quickly. It was not possible to separate live trees from dead trees in estimating the pre-fire loadings of the fraction. However, the portion represented by dead trees was assumed to be negligible because the occurrence of dead trees (snags) in cutover southwestern ponderosa pine forests such as the pre-fire forests on the Stermer Ridge watersheds before the wildfire was generally insignificant (Ffolliott 1983a).

Downed Woody Materials

Loadings of downed woody materials consisting of the accumulations of stems, branches, and twigs on the forest floor before the wildfire were estimated by the value for this fraction reported earlier by Sackett (1979) and Sackett and Haase (1996). Measurements of downed woody materials included in this value were obtained on 14 sites in the vicinity of the Stermer Ridge watersheds. The post-fire loadings of this fraction on the Stermer Ridge watersheds were estimated by applying the planar-intersect procedure (Brown 1974; Brown and others 1982) at the plot. This procedure involved counting the intersections of stems, branches, and twigs on the forest floor with a vertical sampling frame that resembles a guillotine that is dropped through the accumulated fuels. The procedure has the same theoretical basis as the line-intersect method of sampling flammable fuels (Van Wagner 1968). Converting the counts obtained with the planar-intersect procedure to estimates of weight is outlined in the publications cited for this procedure. The standing trees killed outright by the fire or severely damaged by the burn that had fallen to the ground were included in the post-fire estimates of this fraction. Estimates of the downed woody materials were obtained in 2003 and 2005 to assess the impacts of the wildfire on the loadings of the fraction. An estimate of this fraction was also obtained after the salvage cutting and fuel reduction treatments were applied to Watershed A to evaluate the impacts of the treatments.

Litter and Duff Accumulations

Obtaining estimates of the pre-fire loadings of fine fuels consisting of litter and duff accumulations on the forest floor involved two steps. A derived ratio of annual litter fall (Davis and others 1968) to annual litter decomposition (Klemmedson and others 1985) was used in projecting the depths of litter and duff layers reported earlier by Ffolliott and others (1976) to estimates of corresponding depths at the time of the wildfire. These estimates of depths were then converted to estimates of weights by applying the appropriate density values (Ffolliott and others 1976) to represent the loadings of this fraction. Post-fire estimates of the fraction were obtained in 2003 and 2005 by the planar-intersect procedure (Brown 1974; Brown and others 1982).

Herbaceous Plants and Shrubs

Production of herbaceous plants and shrubs reported in the earlier studies on the watersheds (Ffolliott and Baker 1977) represented the pre-fire estimates of this fine fuel fraction. Production estimates that were obtained at the lower end of the annual precipitation regimes encompassed in the earlier studies were selected as the bases for comparison with the post-fire estimates of the production of the plants obtained on the watersheds during the drought conditions that prevailed throughout this study. Because estimates of only the late-growing herbaceous plants and shrubs were available from the earlier studies, the estimates of the production of late-growing plants obtained in 2003 and 2005 represented the post-fire loadings of this fraction.

Statistical analyses of differences in the estimated loadings of flammable fuels on the Stermer Ridge watersheds were not undertaken. The varying measurement methods and sampling procedures involved in estimating the pre- and post-fire loadings of the respective fractions precluded these analyses. Only general statements on the pre- and post-fire comparisons, therefore, have been made.

Impacts on Ecosystem Resources

Tree Overstories

Ponderosa Pine Trees

Nearly 50 percent of the ponderosa pine trees on Watershed A that were burned by high severity fire were killed outright by the wildfire (fig. 7A).

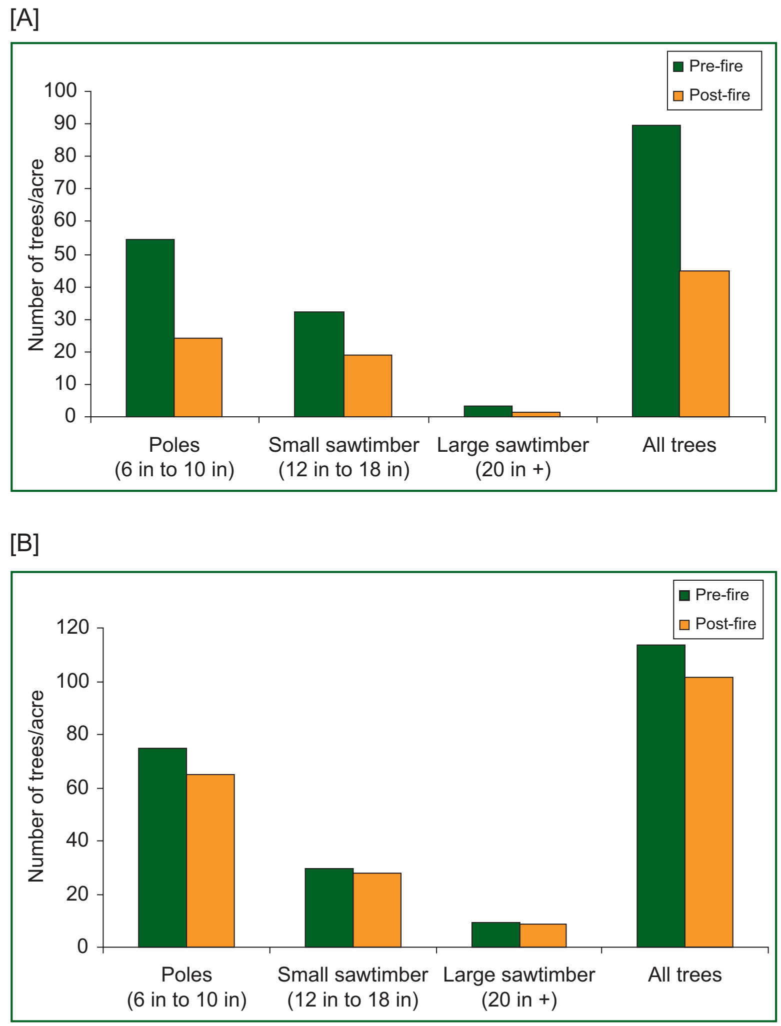


Figure 7—Initial mortality of the Rodeo-Chediski Wildfire on the overstories of ponderosa pine trees on the Stermer Ridge watersheds burned by (A) a high severity fire and (B) a low severity fire in terms of number of trees per acre. Scales of the Y-axis are unique for each graph.

There was higher mortality in the pole class than the sawtimber classes of trees, a common finding following wildfires in coniferous forests of the western United States (Arno 2000). Almost 55 percent of the poles suffered immediate mortality, while about 35 percent of the trees in the sawtimber classes were killed. All of the trees that survived the initial impact of the burn were severely damaged, with more than two-thirds of each tree's crown scorched. Almost two-thirds of those trees died within two years. Continuing rates of tree mortality averaged nearly 5 percent annually until the salvage cutting and fuel reduction treatments were implemented on the watershed.

Only 17 percent of the ponderosa pine trees on Watershed B were killed in the burn due low fire severity (fig. 7B). The pole class once again suffered the greatest mortality, while mortality in the sawtimber classes was less than 10 percent. Of the trees that survived the wildfire, 70 percent were undamaged by the burn, and most of the remaining trees exhibited only limited crown damage. Less than 3 percent of the fire-damaged trees that survived the initial impact of the wildfire died by 2004, with only minimal tree mortality observed thereafter. Post-fire mortality of trees on this watershed was similar to that reported by O'Brien (2002) for unburned southwestern ponderosa pine forests.

Impacts of the Rodeo-Chediski Wildfire in terms of basal area and volume losses of ponderosa pine trees have been reported in a more comprehensive paper on the effects of the wildfire on the tree overstories of the Stermer Ridge watershed by Ffolliott and others (2008) and, therefore, are not detailed here. Very briefly, however, almost 45 percent of both the basal area and volume of the ponderosa pine trees on Watershed A were lost initially to the wildfire. The largest losses of basal area occurred in the large sawtimber class, while trees in the pole class suffered the largest volume losses. Basal area and volume losses of ponderosa pine trees on Watershed B were minimal.

Other Tree Species

Almost 90 percent of the Gambel oak trees on Watershed A suffered complete crown-kill as a result of the high severity fire. However, basal sprouting was observed around most of these trees within one year of the wildfire, indicating that many of the trees had not been root-killed by the burn. Nearly all of the Gambel oak trees on Watershed B survived the wildfire with little or no fire damage. While none were tallied by point sampling on either of the Stermer Ridge watersheds, the few alligator juniper trees observed on Watershed A were either killed by the wildfire or suffered severe crown damage. Only limited post-fire basal sprouting of alligator juniper occurred. Most of the alligator juniper on the Watershed B were not affected by the burn.

Other Studies on Impacts of Wildfire on Tree Overstories

Similar impacts of wildfire on tree overstories have been reported in coniferous forests of the western United States in general (Arno 2000) and, more specifically, other southwestern ponderosa pine forests (Campbell and others 1977; Dieterich 1979; Fowler and Sieg 2004; Harrington 1987; McHugh and Kolb 2003; Pearson and others 1972). These studies collectively show that the effects of a wildfire on the tree overstories impacted are determined largely by the pre-fire stand structures of the burned forests and severity of the wildfire. Monitoring impacts of the Rodeo-Chediski Wildfire on the tree overstories of the Stermer Ridge watersheds represents one of the more continuous and longer-term evaluations of the post-fire impacts of wildfire on tree overstories in the Southwestern Region.

Stocking of Ponderosa Pine Seedlings

Impacts of Fire Severity

Only two of the 30 mil-acre plots on Watershed A were stocked with ponderosa pine seedlings that were alive before the wildfire in the fall of 2002, while no plots were stocked with seedlings that had germinated after the fire. This latter finding was not expected, however, because of the large number of newly germinated ponderosa pine seedlings that were observed on the watershed shortly after the wildfire. No plots were stocked with either pre- or post-fire seedlings in 2004 or 2006, indicating that the post-fire environmental conditions on this watershed were (apparently) not conducive to the survival of ponderosa pine seedlings in the immediate aftermath of the burn.

Ponderosa pine seedlings that were presumed to be alive before the wildfire stocked one-half of the 30 mil-acre plots on Watershed B in the fall of 2002, although some of the plantlets had been scorched. However, the condition of the seedlings at that time indicated that many had also suffered from the prevailing drought conditions. It was not surprising, therefore, that only two of the plots were stocked with pre-fire seedlings in 2004 and none was stocked with seedlings in 2006. No plots were stocked at any time with ponderosa pine seedlings that had germinated after the wildfire.

Stocking of ponderosa pine seedlings after the Rodeo-Chediski Wildfire was insufficient in both number and distribution to maintain the forest at a stocking level necessary to ensure for future sustainability (Pearson 1950; Schmidt 1988; Schubert 1974; Tecle and Hamre 1989; and others) on either of the Stermer Ridge watersheds. The prolonged drought that prevailed at the time of the wildfire and thereafter also impacted obtaining and then sustaining post-fire natural reproduction. Moreover, natural reproduction of ponderosa pine in the Southwestern Region is episodic in nature (Pearson 1950; Schubert 1974), which must also be considered in the assessment of post-fire stocking conditions.

Other Studies on Impacts of Wildfire on Stocking of Ponderosa Pine Seedlings

Campbell and others (1977) found no ponderosa pine seedlings in their three-year evaluation of the impacts of a wildfire in a southwestern ponderosa pine forest. However, the impacts of that wildfire on natural reproduction were inclusive because there also was a lack of established seedlings on adjacent unburned sites. In comparing the stocking conditions of ponderosa pine seedlings on four sites that had burned within 20 years of each other near Flagstaff, Arizona, with an unburned area, Lowe and others (1978) reported that ponderosa pine seedlings were found on all of the burned sites. Peak tallies of these seedlings occurred within two years of a wildfire event and then leveled off but still remained higher than the stocking of seedlings on the unburned site. To place this finding into proper perspective, the annual precipitation amounts throughout the 20-year study period were above average.

Species Compositions of Herbaceous Plants and Shrubs

Among the herbaceous plants and shrubs affected by high severity fire on Watershed A was an assemblage of grasses dominated by squirreltail, muttongrass, blue grama, and mountain muhly. Other grasses of less common occurrence were prairie Junegrass, Arizona fescue, and black dropseed. Mullein (*Verbascum thapsus*), a short-lived biennial plant naturalized from Europe (Epple 1995; Sieg and others 2003), dominated the forb component of the herbaceous plants on both of the Stermer Ridge watersheds in the early years after

Figure 8—Mullein (*Verbascum thapsus*) was the main forb component of herbaceous plants on both of the Stermer Ridge watersheds in the early years following the wildfire. (*Photo by Peter F. Ffolliott.*)

the wildfire (fig. 8). Cooley's bundleflower, Cuman's ragweed, and western yarrow replaced mullein in the latter years of the study. The limited occurrences of post-fire shrubs on the watersheds were mostly basal sprouts from the crown-killed Gambel oak trees with scattered Fendler's ceanothus.

The post-fire composition of herbaceous plants and shrubs on Watershed B was similar to the pre-fire composition (Ffolliott and Baker 1977). The frequent occurrence of mullein in the early years of the study was the main exception.

Production of Herbaceous Plants and Shrubs

Production of herbaceous plants and the grass and forb components of the herbaceous plants are illustrated in this paper by bar graphs that show the respective production of these plant groups in the sampling periods (see figs. 9, 10, and 11). Because the data sets that formed the basis for constructing the bar graphs were mostly non-normal in their frequency distributions, however, inferences relating to the apparent differences among the levels of production are not necessarily valid. Occurrences of statistical differences in production in the plant groups were determined through interpretations of the results that were obtained from the non-parametric tests applied in the study.

Herbaceous Plants

Production of herbaceous plants was generally greater throughout the study period on Watershed A than on Watershed B where fire severity was low (fig. 9). The only exception to this finding occurred in the fall of 2002, shortly after cessation of the wildfire, when the low levels of herbage production that were estimated on the two watersheds were similar. The situation was probably a consequence of the initial impacts of the wildfire. Infiltration of water into the soil at that time was inhibited by the formation of water-repellent soils on the two watersheds (Stropki and others 2005) that, in turn, limited the amount of soil moisture available for plant growth. Though not measured in this study, the possibility of accelerated losses of on-site nutrients in the initially high overland flows of water that occurred shortly after the wildfire (Garcia-Chevesich and others 2004) could have been another causal factor.

The greater production of herbaceous plants for most of the study period on Watershed A was attributed largely to the greater loss of the tree overstory on this watershed than on the other watershed. Earlier studies in southwestern ponderosa pine forests have shown that the production of herbaceous plants increases when densities of the competing tree overstory decrease (Bojorquez-Tapia and others 1990; Clary 1975; Ffolliott 1983b; and others). Burning intensities that consume the litter and large amounts of the duff layers can also result in increased herbage production in these forests (Clary and others 1968; Davis and others 1968). Such a fire occurred on the severely burned watershed.

Another reason for the post-fire pattern of production of the herbaceous plants on the severely burned watershed could be that a large pool of available nutrients was formed where large quantities of vegetation, litter, and duff were combusted by the fire. The trend of the post-fire production illustrated in fig. 9 agrees with the effects that one might anticipate from a flush of available nutrients following fire (Bond and van Wilgen 1996; DeBano and others 1998; Whelan 1997). While production of the herbaceous plants in the fall of 2002 and the spring of 2003 might not have been affected by a post-fire nutrient pulse, an increase in the production could be expected following the summer rains in 2003. Such an increase would likely be sustained for a couple of years and then would start to decline along with the nutrients made available by the wildfire. The increase in production of herbaceous plants on Watershed B was less because a smaller pool of nutrients was released.

Production of early-growing herbaceous plants was less each year than the production of late-growing herbaceous plants on Watershed A, with the exception of 2005, when production of the two groups of plant was similar (fig. 9). However, comparisons of the differences in production of early-growing plants to the production of late-growing

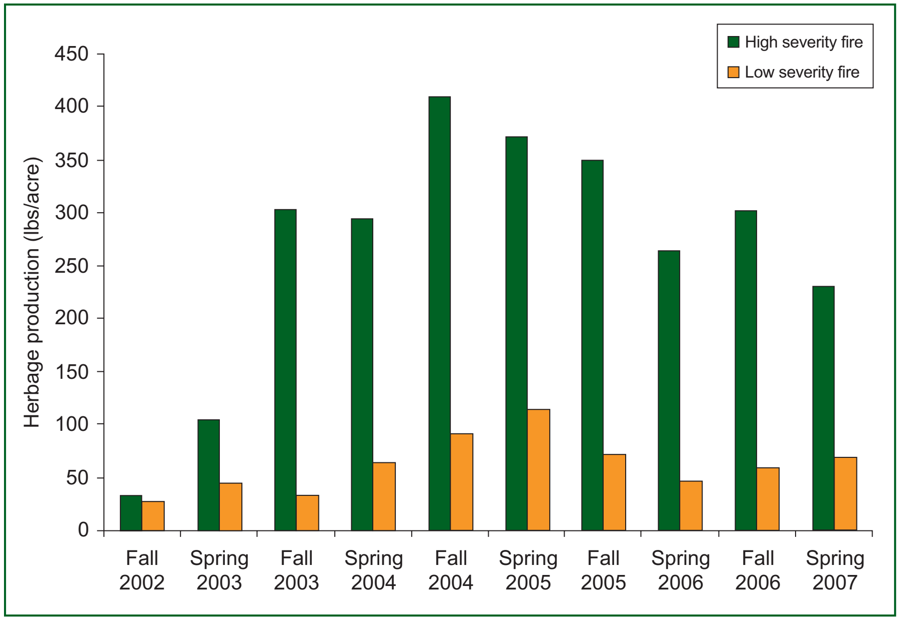


Figure 9—Production of herbaceous plants on the Stermer Ridge watersheds after the Rodeo-Chediski Wildfire. Estimates of average production are shown in this figure.

Table 2—Differences in the production of early-growing herbaceous plants and in the production of late-growing herbaceous plants (considered separately) on the Stermer Ridge watersheds that were burned by a high severity fire and a low severity fire. Higher levels of production of herbaceous plants are indicated by the years to the left on the table with lower levels of production progressing to the right. Years not underscored by the same line are significantly different, while years underscored by the same line are not significantly different.

High severity fire					
Early-growing herbage	Spring 2005	Spring 2004	Spring 2006	Spring 2007	Spring 2003
Late-growing herbage	Fall 2004	Fall 2005	Fall 2003	Fall 2006	Fall 2002
Low severity fire					
Early-growing herbage	Spring 2005	Spring 2007	Spring 2006	Spring 2004	Spring 2003
Late-growing herbage	Fall 2005	Fall 2004	Fall 2006	Fall 2003	Fall 2002

plants on Watershed B were inconsistent, with production of early-growing plants greater than that of late-growing plants in some years and vice versa in other years.

There were significant but inconsistent differences in the production of early-growing herbaceous plants on the two watersheds throughout the study (table 2). While there were significant differences in the production of late-growing plants on the watersheds, the differences were also inconsistent. The time of recovery since the Rodeo-Chediski Wildfire occurred and the seasonal precipitation amounts that affected the production of the respective plant groups were evaluated in attempting to explain these inconsistencies. However, the time since the wildfire occurred was not a significant contributing factor. But the production of the two plant groups was favorably related to the seasonal precipitation amounts on one of the two watersheds.

Production of early-growing and late-growing herbaceous plants was not related to the seasonal precipitation amounts on Watershed A. The more severe impacts of the Rodeo-Chediski Wildfire on ecosystem functioning of this watershed may have obscured any influence that seasonal precipitation might have had on the production of herbaceous plants. However, the production of both early-growing and late-growing plants on Watershed B related linearly to the amount of seasonal precipitation. On this watershed, the correlation coefficient between the production of early-growing plants and precipitation between October 15 and April 14 and the correlation coefficient between the production of late-growing plants and precipitation between April 15 and October 14 were significant, similar in magnitude, and, therefore, combined. The resulting correlation coefficient (+0.748) indicated that 56 percent of the variation in the production of both early-growing and late-growing herbaceous plants on Watershed B could be attributed to the precipitation amount that affected the production of herbaceous plants for that season.

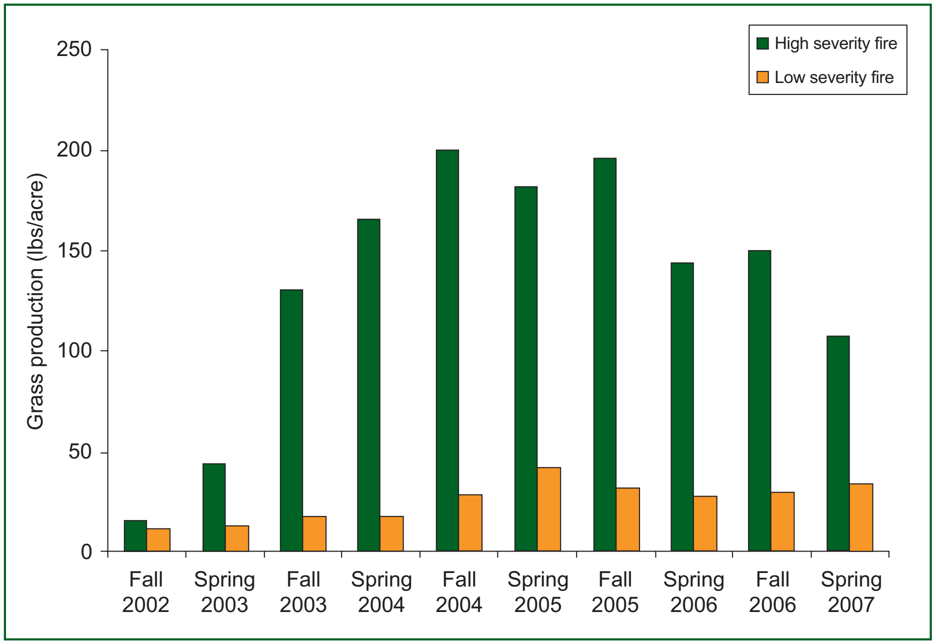


Figure 10—Production of the grass component of herbaceous plants on the Stermer Ridge watersheds after the Rodeo-Chediski Wildfire. Estimates of average production are shown in this figure.

Table 3—Differences in the production of early-growing grasses and in the production of late-growing grasses (considered separately) on the Stermer Ridge watersheds that were burned by a high severity fire and a low severity fire. Higher levels of production of grasses are indicated by the years to the left on the table with lower levels of production progressing to the right. Years not underscored by the same line are significantly different, while years underscored by the same line are not significantly different.

High severity fire

Early-growing grasses

Spring 2005	Spring 2004	Spring 2006	Spring 2007	Spring 2003

Late-growing grasses

Fall 2005	Fall 2004	Fall 2006	Fall 2003	Fall 2002

Low severity fire

Early-growing grasses

Spring 2005	Spring 2007	Spring 2006	Spring 2004	Spring 2003

Late-growing grasses

Fall 2005	Fall 2006	Fall 2004	Fall 2003	Fall 2002

Grasses

Grass production was also greater on Watershed A despite high severity fire, with the exception of the fall shortly after the wildfire when the levels of production were similar (fig. 10). The greater loss of the tree overstory, more complete consumption of the litter and duff layers, and (possibly) greater pool of available nutrients on this watershed were among the factors assumed to have contributed to the difference.

Production of early-growing grasses was less than the production of late-growing grasses on Watershed A throughout the study, with the exception of 2006 when the production of the two groups of grasses was similar (fig. 10). Production of early-growing grasses was also less than that of the late-growing grass for most of the study period on Watershed B. An exception to this finding occurred in 2005, when the levels of production for the respective grasses were similar for unknown reasons.

Production of early-growing grasses on Watershed A in 2003 was significantly lower from the production of early-growing grasses in the other years of the study for unknown reasons (table 3). There also were significant differences in the production of late-growing plants on this watershed, but the pattern of production of these plants was inconsistent. Production of early-growing grasses on Watershed B differed inconsistently, while the production of late-growing grasses was similar throughout the study. The time since the wildfire was not a factor in that contributed to those differences. However, the seasonal amount of precipitation that regulated the production of the two groups of grasses was a significant factor on Watershed B.

There was no relationship between either the production of early-growing grasses or the production of late-growing grasses and the seasonal precipitation amounts on Watershed A. But the production of early-growing and late-growing grasses was related linearly to the amount of seasonal precipitation associated with their production levels on Watershed B. The correlation coefficient between the production of the early-growing grasses and precipitation amounts for October 15 to April 14 and the correlation coefficient between the production of the late-growing grasses and precipitation between April 15 and October 14

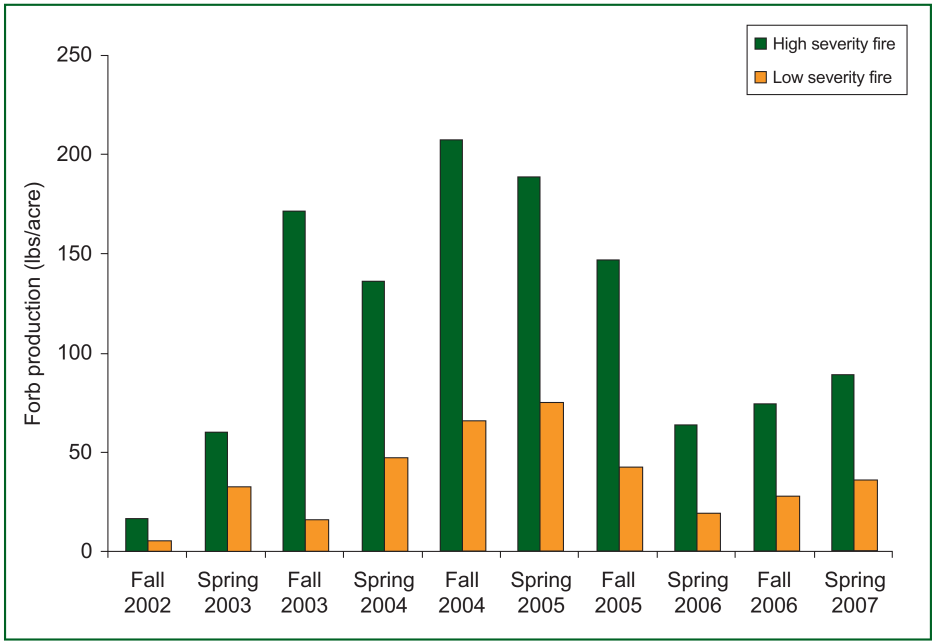


Figure 11—Production of the forb component of herbaceous plants on the Stermer Ridge watersheds after the Rodeo-Chediski Wildfire. Estimates of average production are shown in this figure.

Table 4—Differences in the production of early-growing forbs and the production of late-growing forbs (considered separately) on the Stermer Ridge watersheds that were burned by a high severity fire and a low severity fire. Higher levels of production of forbs are indicated by the years to the left on the table with lower levels of production progressing to the right. Years not underscored by the same line are significantly different, while years underscored by the same line are not significantly different.

High severity fire					
Early-growing forbs					
	Spring 2005	Spring 2004	Spring 2007	Spring 2006	Spring 2003
Late-growing forbs					
	Fall 2003	Fall 2004	Fall 2005	Fall 2006	Fall 2002
Low severity fire					
Early-growing forbs					
	Spring 2005	Spring 2004	Spring 2007	Spring 2006	Spring 2003
Late-growing forbs					
	Fall 2004	Fall 2005	Fall 2006	Fall 2003	Fall 2002

were significant, similar, and combined. The resulting correlation coefficient (+0.686) indicated that 47 percent of the variation in the production of grasses could be attributed to the amount of precipitation that influenced grass growth in that season.

Forbs

Production of forbs was consistently higher throughout the study on Watershed A than on Watershed B (fig. 11). The greater reduction of the tree overstory, more complete consumption of the litter and duff layers, and availability of nutrients on Watershed A were again the probable reasons for this finding.

Production of early-growing forbs was less than the production of late-growing forbs throughout the study on Watershed A, with the exception of 2005 when the two levels of production were similar (fig. 11). However, the differences in production of the two groups of forbs on Watershed B were inconsistent, with similar levels of production in 2003 and 2006 but different levels of production in 2004 and 2005.

Differences in the production of early-growing forbs were insignificant on both of the watersheds throughout the study period. However, the production of late-growing forbs differed significantly but inconsistently on the two watersheds (table 4). The time since the wildfire occurred was not related to the production of the two groups of forbs. Seasonal precipitation amounts were related to the level of production of the two groups on one of the watersheds but not on the other.

Neither the production of early-growing nor late-growing forbs on Watershed A was correlated with the influencing seasonal precipitation amounts. However, the production of these forb groups was related linearly to the amount of seasonal precipitation linked to their production on Watershed B. The correlation coefficient between production of the early-growing forbs and seasonal precipitation amounts and the correlation coefficient between production of the late-growing forbs and seasonal precipitation were significant, similar, and combined. The resulting correlation coefficient (+0.693) indicated that 48 percent of the variation in the production of forbs was attributed to seasonal precipitation. This amount of accountable variation was similar to the amount associated with the production of grasses and seasonal precipitation.

Mullein

Mullein often establishes itself on a site in southwestern ponderosa pine forests following a fire (Epple 1995; Ffolliott and others 1977; Sackett and others 1994; Sieg and others 2003). This was also the case on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire. This early-seral species was found on 25 percent of the 9.6-ft^2 plots and contributed nearly 50 percent to the production of the forb component at its peak establishment on Watershed A in the fall of 2004 (fig. 12A). Mullein occurred on 10 percent of the plots and represented about 40 percent of the comparatively lower level of forb production at its peak on Watershed B in the following spring (fig. 12B). The early dominance of the plant declined on both of the watersheds thereafter.

Mullein has been reported to be a major component of the early production of herbaceous plants in other studies on the effects of wildfire in southwestern ponderosa pine forests (Davis and others 1968; Ffolliott and others 1977; Sackett and others 1994). The plant rarely becomes overly aggressive, however, because bare soil is required for its seeds to germinate (Sackett and others 1994; Sieg and others 2003). Mullein is intolerant of the shade from competing herbaceous plants. It was expected, therefore, that the species would decline in its occurrence and level of production after peaking on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire as the amount of bare ground and availability of light decreased and other herbaceous species recovered from the burn.

Shrubs

The infrequent post-fire occurrence of shrubs on Watershed A contributed little to the production of understory plants (less than 10 lbs/acre), and this limited amount of production occurred in only the spring and fall of 2006 and the spring of 2007. Basal sprouts of crown-killed Gambel oak trees comprised most of this understory component. Only a trace of shrub

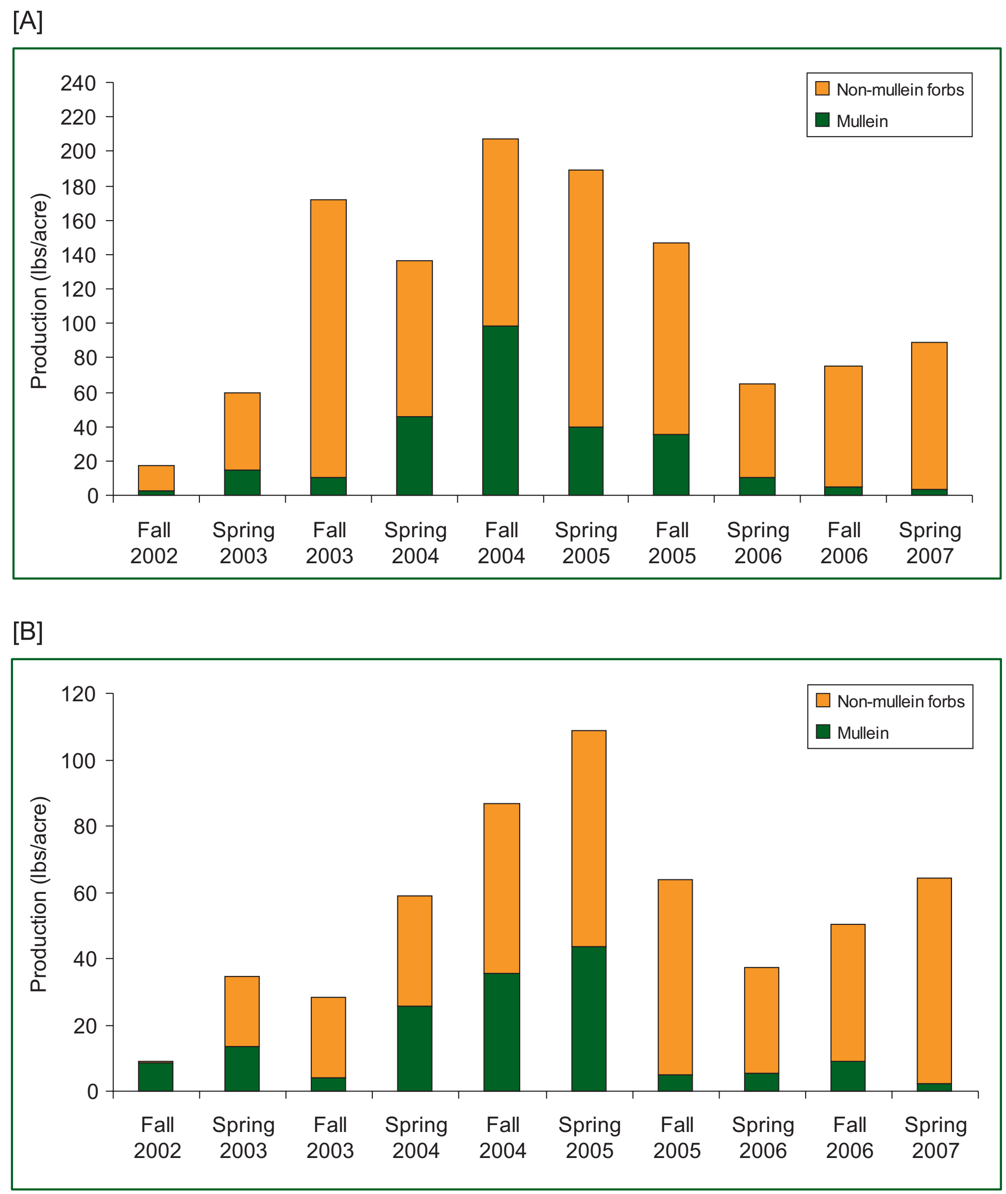


Figure 12—Proportions of the forb component of herbage represented by mullein (*Verbascum thapsus*) on the Stermer Ridge watersheds burned by (A) a high severity fire and (B) a low severity fire. Scales of the Y-axis are unique for each graph.

production occurred on Watershed B even though it was burned by a low severity fire. Otherwise, the shrub production was insignificant on this watershed throughout the study period.

Other Studies on Impacts of Wildfire on Species Compositions and Production of Herbaceous Plants and Shrubs

Impacts of wildfire on herbaceous plants and shrubs in the coniferous forests of the western United States have been reported generally by Arno (2000) and, more specifically, in southwestern ponderosa pine forests by Campbell and others (1977), Lowe and others (1978), Oswald and Covington (1983), Pearson and others (1972), and others. In reference to the impacts of fire on species compositions, a shift from a largely mullein composition in the initial years following fire to other herbaceous species as post-fire time progresses has also been found in other studies in southwestern ponderosa pine forests (Davis and others 1968; Ffolliott and others 1977). Results also show that compositions of herbaceous plants and shrubs remain mostly unchanged after mullein declines in its post-fire occurrence.

In a study similar to that on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire, Campbell and others (1977) estimated the production of late-growing herbaceous plants and shrubs on three watersheds in north-central Arizona that had been (respectively) severely burned, moderately burned, and unburned by a wildfire. The basis for classifying the fire severity of the watersheds was not specified by Campbell and others (1977), but it was not the classification system of Hungerford (1996) that evolved later. The researchers found that the levels of plant production on the three watersheds six months after the wildfire were not different—a finding not unlike that obtained on the Stermer Ridge watersheds in the fall of 2002 shortly after the Rodeo-Chediski Wildfire. However, the production of herbaceous plants and shrubs on the moderately and severely burned watersheds increased nearly three-fold 30 months later in comparison to that estimated on the unburned watershed. Similar results of increased herbage production were also observed on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire, with the largest increases in herbage production on Watershed A.

Utilization of Forage Plants

Only limited amounts of forage plants were utilized by herbivores in the study period. Cattle had been removed by local ranchers from the Stermer Ridge watersheds and, more generally, from the larger area burned by the Rodeo-Chediski Wildfire when the magnitude of the fire became evident. A small (but unknown) number of cattle were allowed back onto the watersheds and nearby vicinity in the summer of 2005. Utilization of forage by the indigenous ungulates was variable throughout the study. The overall utilization of forage species by all of the herbivores averaged 3 percent annually on a watershed basis.

Indigenous Ungulates and Small Mammals

Information on the elk and mule deer populations in the Stermer Ridge watersheds before the Rodeo-Chediski Wildfire was incomplete. It was known, however, that an “exceptionally large” elk population inhabited the general area before the wildfire, while the mule deer population was below the “long-term norm” for the area (U.S. Forest Service 2002). Whether these two populations of indigenous ungulates were expanding, declining, or stable was unknown. Little was known about the small mammal populations on the watersheds.

Elk

The initial disruptive impact of the Rodeo-Chediski Wildfire was the assumed cause of the limited and similar presence of elk on the two Stermer Ridge watersheds in the fall of 2002 (fig. 13). However, in the

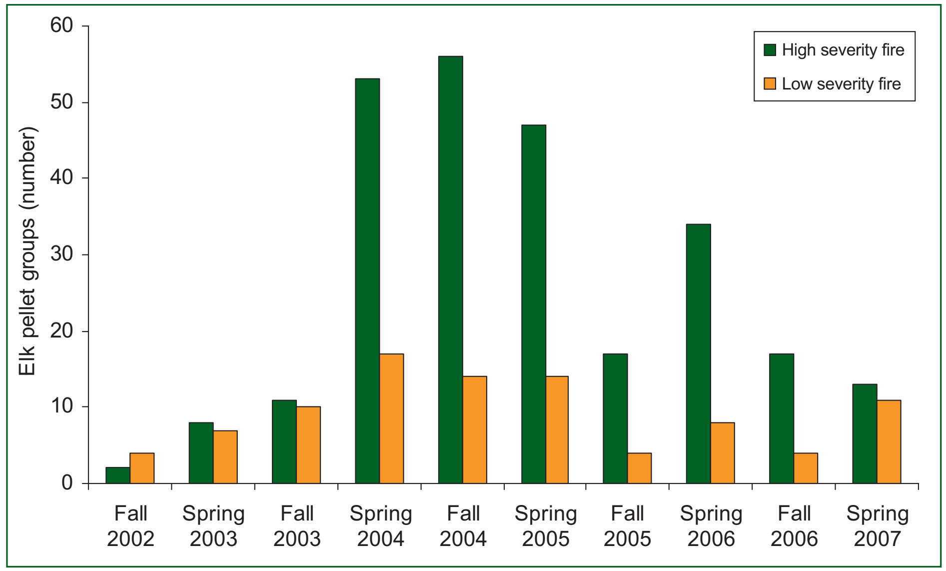


Figure 13—Presence of elk (*Cervus elaphus*) on the Stermer Ridge watersheds after the Rodeo-Chediski Wildfire, as indicated by indicated by pellet group counts. Total pellet group counts in the respective sampling periods on the watersheds are shown in this figure.

spring of 2004, the occurrence of elk on Watershed A increased significantly in relation to the occurrence observed on Watershed B. This difference continued through the following spring, after which the post-fire presence of elk on the two watersheds was similar to the end of the study.

It was assumed that the increase in the production of forage plants preferred by elk (Clary and Larson 1971; Kufeld 1973; Thill and others 1983) on Watershed A in the middle years of the study contributed to the comparatively high presence of elk on the watershed at that time. Included among the preferred forage plants were muttongrass, squirreltail, Cooley's bundleflower, and Fender's ceanothus. Availability of the protective cover provided by standing trees at that time was possibly a secondary factor of importance. While the production of the preferred forage plants was sustained at a relatively high level to the end of the study, the cover of standing trees killed outright or severely damaged by the Rodeo-Chediski Wildfire started to decline by the summer of 2005 when those trees started falling to the ground. The loss of standing trees increased further into 2006 when the salvage cutting and fuel reduction treatments were implemented. Disturbances caused by the activities associated with post-fire interventions such as the felling, skidding, and piling of the larger trees might have also contributed to the declining presence of elk on this watershed.

The lower occurrences of elk on Watershed B remained largely unchanged throughout most of the study (fig. 13). Availability of preferred forage species and protective cover on this watershed was impacted less by the wildfire. Also, salvage cutting and fuel reduction treatments were not imposed on this watershed.

Elk populations in northern Arizona generally migrate from their summer habitats in the higher mountains to winter habitats at the lower elevations. However, because of the low snow accumulations and above-average temperatures in the winters of the study, many elk remained on or near the Stermer Ridge watersheds throughout the year, as indicated by the seasonal pellet group counts.

Mule Deer

The presence of mule deer on the two Stermer Ridge watersheds was less than the elk presence throughout the study. Whether the lower occurrences were a reflection of the small mule deer population in the area before the wildfire or a consequence of the burn lowering the habitat qualities for the ungulate on the watersheds is unknown. Within the perspective of their lower occurrence on the Stermer Ridge watersheds, the presence of mule deer in the fall of 2004 and spring of 2005 on Watershed A was greater than the occurrences on Watershed B (fig. 14). It should be noted that no mule deer pellet groups were counted on Watershed B in the fall of 2004, the fall of 2005, and the spring of 2007.

The greater presence of mule deer in the fall of 2004 and the spring of 2005 on Watershed A was attributed largely to the relatively short-term increases in production of the browse species preferred by mule deer (Clary and Larson 1971; Neff 1974; Severson 1981; Thill and others 1983) during those times compared to the decreased production of the species in other sampling periods. The basal sprouts that originated from crown-killed Gambel oak trees in the later stages of this study were the only browse of significance on either of the watersheds after the wildfire. The protective cover of the standing trees had little apparent impact on the movements of mule deer.

Mule deer populations also remained in the general area of the Stermer Ridge watersheds throughout most of the study period, as suggested by the seasonal pellet group counts. The limited snow accumulations and above average temperatures apparently precluded the need for mule deer to move to habitats at lower elevations in the winter.

Small Mammals

It was assumed that most of the cottontail and Abert's squirrels that inhabited the Stermer Ridge watersheds were unable to escape the Rodeo-Chediski

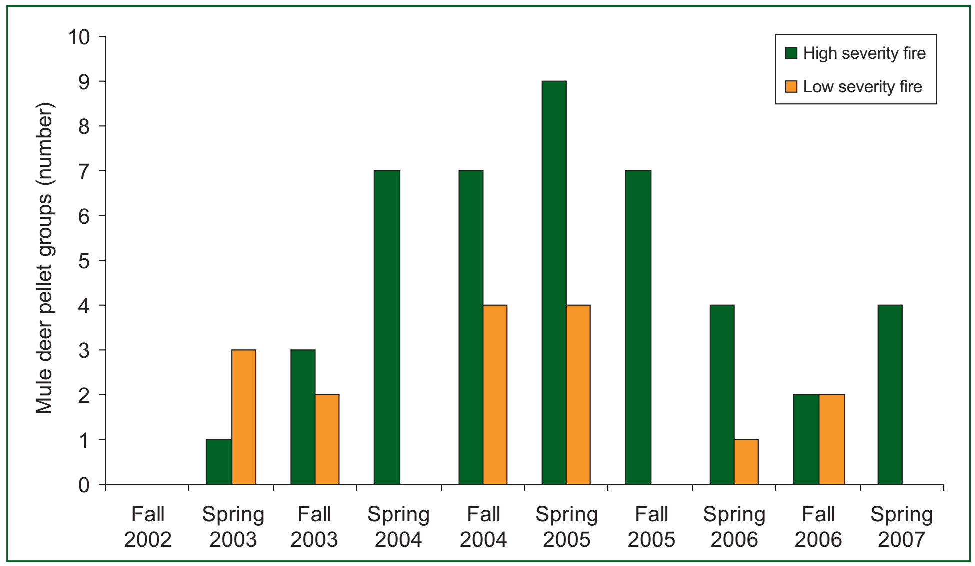


Figure 14—Presence of mule deer (*Odocoileus hemionus*) on the Stermer Ridge watersheds after the Rodeo-Chediski Wildfire, as indicated by pellet group counts. Total counts of pellet groups in the respective sampling periods on the watersheds are shown in this figure. (The scale of the Y-axis in this figure differs from the scale of the Y-axis in fig. 13 for elk.)

Wildfire because of their small home ranges (Brown 1984; Patton 1975b; U.S. Natural Resources Conservation Service 1999) and the extensive area of the burn. Therefore, the limited presence of cottontail in the area was expected. Insufficient food sources (U.S. Natural Resources Conservation Service 1999) and a lack of sufficient cover conditions (Costa and others 1976) for most of the study were the probable reasons for the low cottontail presence on either of the watersheds after the wildfire.

Abert's squirrels were not observed throughout the study period on Watershed A due to the high severity fire. Most of the feed trees preferred by squirrel (Brown 1984; Ffolliott and Patton 1978; Keith 1965; Patton 1975a) had been killed by the wildfire or died shortly thereafter. As a consequence, this key habitat requirement was lost in the burn. However, following their initial displacement on Watershed B, scattered twig clippings indicated that the squirrel was feeding on this watershed two years after the wildfire. Many of the feed trees preferred by the squirrel had survived the burn and, therefore, were available to the returning small mammal.

Other Studies on Impacts of Wildfire on Mammalians

Impacts of wildfire on the ungulates and small mammals that inhabit southwestern ponderosa pine forests have long been a concern of wildlife managers in the Region. It is hoped, therefore, that the impacts of the Rodeo-Chediski Wildfire on these wildlife groups reported in this paper will contribute to the existing information already available on wildfire impacts on wildlife (Campbell and others 1977; Clary and Larson 1971; Finch and others 1997; Kruse 1972; Kufeld 1973; Lowe and others 1978; Thill and others 1983). This information should add to the database on the effects of wildfire on the mammalians inhabiting the coniferous forests of the western United States (Smith 2000). The results obtained on the Stermer Ridge watersheds, combined with the conclusions from earlier studies, indicate that the impacts of a wildfire in these forests are usually greater on mammal habitats than on mammal mortality. Small mammals with limited mobility are most vulnerable to a wildfire, while mammals with comparatively large home ranges are more likely to survive a wildfire, responding negatively or positively to changes in habitat conditions depending on the severity, uniformity, and duration of the fire and the extent of the area burned. Changes in the availability of food and amount of protective cover for these animals were especially notable in the stand-replacing, high severity wildfire.

Birds

The number of species and total numbers of birds tallied on the Stermer Ridge watersheds after the wildfire were less than counts generally reported in unburned southwestern ponderosa pine forests (Brawn and others 1987; Hurteau and others 2008; Rosenstock 1996; Szaro and Balda 1979). The widespread impacts of the Rodeo-Chediski Wildfire, when coupled with the ongoing drought in the Southwestern Region, probably contributed to the limited numbers of birds sighted. Birds in the foliage-gleaning guild of species (Ehrlich and others 1988) declined in numbers because of the loss of food items such as invertebrates and fruit on the foliage and branches of trees killed by the wildfire (fig. 15). Birds whose nests are supported by trees also declined on the severely burned sites. However, birds that were nested in the cavities of standing dead trees benefited from the fire until the trees fell to the ground. Birds scavenging food from the ground surface were likely affected less by the wildfire.

Figure 15—Birds in the foliage-gleaning guild of species likely declined in numbers on severely burned sites on the Stermer Ridge watersheds because of the loss of food items such as invertebrates and fruit on the foliage and branches on trees killed by the wildfire. (*Photo by Daniel G. Neary*.)

Impacts of Fire Severity

It was anticipated that the Rodeo-Chediski Wildfire might impact the species and numbers of birds tallied on Watershed A differently than on Watershed B. However, occurrences and summaries of bird species numbers on the two watersheds indicated that the wildfire effects were mostly insignificant or generally inconclusive, with a few exceptions (Sibley 2000). Fewer birds were sighted in the fall of 2002, shortly after cessation of the wildfire and during the following year, on Watershed A. The presumed cause was the devastating direct and indirect impacts of the wildfire. Following this initial shock produced by the wildfire, the differences in the birds tallied on the two watersheds were inconsistent with little meaningful trend.

Species and Numbers of Birds Tallied

Of the 25 species tallied on the two watersheds (combined) in the fall observations, 12 of the species were sighted only once or twice in the study, compared to 10 of the 26 species observed in the spring that were tallied only once or twice. A few other bird species were observed in comparatively large numbers only once on either one or the other of the watersheds. The eight pine siskin (*Carduelis pinus*) individuals sighted in the fall of 2005 on Watershed A were the only observations of this species. The sole observation of a pygmy nuthatch (*Sitta pygmaea*) occurred in the fall of 2005 on Watershed B, when 18 birds were sighted. The 28 juniper titmouse (*Baeolophus ridgwayi*) sighted in the spring of 2007 on Watershed B were the only observations of this species throughout the study.

A few bird species were tallied in comparatively large numbers in only one of the observation periods, with fewer numbers sighted at least once on the watersheds at other times. A large number of hairy woodpecker (*Picoides villosus*) individuals was tallied in the spring of 2007 on Watershed A despite the high severity fire, while fewer were tallied on either watershed at any time either before or after the wildfire. Most of the large numbers of gray-headed junco (*Junco hyemalis*) observed in the study were tallied in the spring of 2005 on Watershed B. The gray-headed juncos were observed later in the fall of 2005 on Watershed A. Bushtit (*Psaltriparus minimus*) was observed in the fall of 2002 on both of the Stermer Ridge watersheds following the wildfire and the following spring. The 20 bushtit sightings on Watershed B represented the largest tally of the species in one observation period. However, this species was not tallied on either of the watersheds for the remainder of the study period.

Seasonal Patterns

Comparisons of the seasonal patterns of species and numbers of birds were possible only in 2003, 2004, and 2005, when tallies of birds were obtained in both the spring and fall. A greater number of species and birds were tallied in the fall than in the spring in each of those years, which possibly reflects the more abundant and diversified food supplies for birds in the summer months (as indicated by the fall tallies) than in the winter (signified by the spring sightings). Some species were tallied at least once in both the fall and spring, although there were a few exceptions when a species was sighted in only one of the two seasons. Pigmy nuthatch and yellow-rumped warbler (*Dendroica coronata*) were tallied in the fall but not the spring on both of the watersheds. Included in spring tallies but not the fall sightings were Grace's warbler (*Dendroica graciae*), juniper titmouse, and violet-green swallow (*Tachycineta thalassina*). Increasing numbers of species and numbers of birds were sighted in both of the seasons as the Stermer Ridge watersheds began to recover from the impacts of the wildfire.

Species Richness, Diversity, and Evenness

Species richness of the birds that were sighted on the Stermer Ridge watersheds was similar in both the spring and fall observations, especially in the latter stages of the study. However, the number of bird species sighted ranged from a low of 3 to a high of 11 for the observation periods in the study, which was fewer than the 12 to 20 species reported by Szaro and Balda (1979) in their study of bird communities in unburned southwestern ponderosa pine forests. Species diversities were generally lower on Watershed A than on Watershed B, with the diversities tending to be lower in the fall than in the spring on both of the watersheds. The species diversities of birds on the Stermer Ridge watersheds were consistently less than the numbers found by Szaro and Balda (1979). Evenness of the bird species observed on the watersheds was variable throughout the study period.

More detailed information on the birds observed on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire may be found in Ffolliott and others (2009). In that document are complete listings of the bird species observed in this study and their numbers tallied by species in the fall and spring observations on both of the watersheds. Comprehensive summaries of the post-fire species richness, diversities, and evenness of birds are also presented.

Other Studies on Impacts of Wildfire on Birds

Less is known about the impacts of wildfire on birds in southwestern ponderosa pine forests than about the effects of wildfire on avifauna in general (Smith 2000), according to a literature review by Finch and others (1997) on the effects of fire, logging, and grazing on bird populations in the Southwestern Region. Interpretations of the results from the few studies cited by Finch and others on the impacts of wildfire on birds in southwestern ponderosa pine forests, including those by Aulenbach and O'Shea-Stone (1983), Blake (1982), Johnson and Wauer (1996), and Lowe and others (1978), were limited, however, because of methodological problems with the studies. Such limitations also occurred in this study of the Rodeo-Chediski Wildfire. For example, species and numbers of birds on the Stermer Ridge watersheds before the wildfire were unknown, which limited a more comprehensive evaluation of the impacts of the burn on the species and numbers of birds tallied. Another limiting factor was that neither replication of the two watersheds studied nor unburned control sites for comparison purposes were available. Furthermore, because the two Stermer Ridge watersheds were relatively small in area, it was likely that some of the birds tallied in this study flew onto one or both of the watersheds from nearby burned sites of differing fire severities or from unburned areas. Therefore, the results presented in this paper should be interpreted within the context of such limitations.

Impacts on Hydrologic Functioning

Water-Repellent Soils

Impacts of Fire Severity

Water-repellent soils can occur at the soil surface as well as in the deeper layers of the soil mantle (DeBano and others 1998; Neary and others 2005). However, the water repellency found on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire was confined mostly to the soil surface (Stropki and others 2005). More specifically, nearly two-thirds of the soils on Watershed A exhibited strong water repellency at the surface immediately after the fire, while moderate water repellency was found on one-third of the soils (fig. 16A). At the same time, strong water repellency was measured at the soil surface on approximately one-third of Watershed B, with 15 percent of the soils having moderate water repellency and almost one-half of the plots with slight or no water repellency (fig. 16B). The post-fire formation of water-repellent soils on both of the watersheds contributed to the record high peak stormflows generated

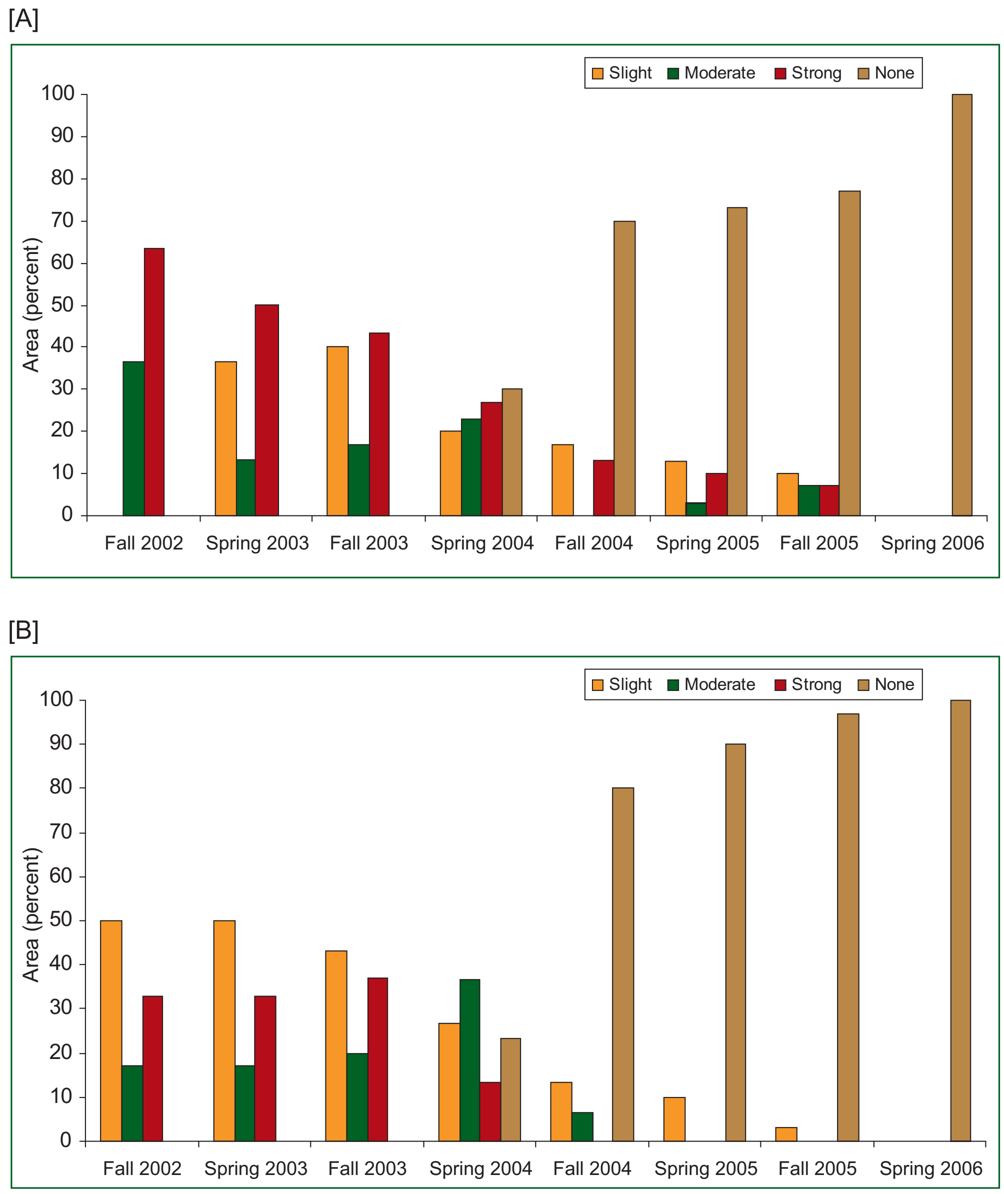


Figure 16—Seasonal occurrences of the water-repellent soils on the Stermer Ridge watersheds that were burned by (A) a high severity fire and (B) a low severity fire. Levels of water repellency shown in this figure are based on criteria of the National Wildfire Coordinating Group (Clark 2001).

by summer rainstorms following the Rodeo-Chediski Wildfire (see the *Peak Stormflows* section).

Most of the water-repellent layers in soils had broken down on both of the Stermer Ridge watersheds within three years of the wildfire (fig. 16). Water repellency was observed three years post-fire on less than 25 percent of Watershed A and only on 10 percent of Watershed B. The decline in the occurrence of water-repellent soils on the two watersheds was attributed largely to the continuing downward erosion of the hydrophobic layer until it had eroded away. Repeated exposures of the soils to a cycle of wetting and drying events also led to a decrease in the degree of water repellency in the soils. Differences in the occurrences of water repellency on the two watersheds, the respective levels of the water repellency, and the patterns of decline in the water-repellent soils between the spring and fall seasons were inconsequential.

Other Studies on Impacts of Wildfire on Water-Repellent Soils

Campbell and others (1977) found that water-repellent soils on a severely burned watershed persisted for three years following a wildfire in a southwestern ponderosa pine forest, while occurrences of water repellency in the soils on a watershed that was burned by a fire of moderate severity was significantly less in this period. Elsewhere in the western United States, Huffman and others (2001) observed water- repellent soils for two years following a wildfire in a ponderosa pine forest on the Front Range of the Rocky Mountains of central Colorado. Dyrness (1976) reported a longer occurrence of water-repellent soils in the soils beneath a lodgepole pine (*Pinus contorta*) forest in the Cascade Mountains of Oregon, where post-fire hydrophobicity persisted for six years. While other researchers have also reported the occurrences of water-repellent soils after a wildfire (DeBano 2000; Neary and others 2005; Erickson and White 2008; National Research Council 2008), only a few have indicated explicitly the duration of post-fire water repellency.

Transpiration and Infiltration

Impacts of Fire Severity

Instantaneous transpiration rates were obtained shortly after the fire on three severely damaged and three undamaged ponderosa pine trees, 18 to 24 inches in dbh, on each of the watersheds. While sap flow was detected in all of the trees, the few repeated measurements (two measurements for each tree sampled) were insufficient to differentiate with confidence the fire-related transpiration losses on the two watersheds. Nevertheless, it was concluded that the loss of water by the transpiration processes was reduced (to unknown levels) on both of the watersheds, with comparatively higher losses on Watershed A because of the greater mortality of trees.

Infiltration of water into the soil was not directly measured in the study. Formation of water-repellent soils in the early years of the study inhibited the infiltration rates on the watersheds, with much of the precipitation that reached the soil surface contributing to increased overland flows. This phenomenon was more pronounced on Watershed A because of the more extensive formation of strong water repellency. Water-repellent soils persisted for three years on the watersheds, after which it was presumed that infiltration rates increased as surface runoff declined.

Other Studies on Impacts of Wildfire on Transpiration and Infiltration

Watershed management studies throughout the world have demonstrated that streamflow can increase following vegetative changes that reduce evapotranspiration losses (Bosch and Hewlett 1982; Troendle and King 1985; Hornbeck and others 1993; Whitehead and Robinson 1993; Zhang and others 2001). Following a wildfire like the Rodeo-Chediski Wildfire, less precipitation is converted into vapor through the transpiration process, and as a consequence, more water is available for streamflow. Vegetation-modifying or vegetation-replacing fire, therefore, can change evapotranspiration (Bond and van Wilgen 1996; Pyne and others 1996; DeBano and others 1998). Fire that modifies the composition and structure of the vegetation by removing foliar volume will result in reduced evapotranspiration losses from a watershed. This is also particularly true for the second fire effect on vegetation case where fire causes a replacement of deep-rooted, high profile trees or shrubs by shallow-rooted, low profile grasses and forbs. This vegetation replacement process leads to reduced transpiration losses because of changes in vegetation characteristics like rooting depth and density. In either instance, reduced evapotranspiration losses following a wildfire usually translate into increased streamflow.

Megahan (1983) reported on increased unsaturated soil water after a wildfire in Idaho. Soil water contents at the end of the first growing season after a wildfire were 44 to 72 percent higher than control areas with intact forest stands due to reduced transpiration of soil water and evaporation from soil surfaces. Shi and others (2009) studied the effects of eucalyptus plantation disturbances on evapotranspiration in China. Wildfire increased water yield mainly due to reductions in transpiration. Some water yield increases were the result of decreased utilization of shallow and deep groundwater by eucalyptus roots. Liu and others (2004) reported on the effects of Australian bushfires on hydrologic processes, including reductions in transpiration at a catchment scale.

Soil Movement

Soil erosion and soil deposition occurred in varying magnitudes at variable times on the hillslopes of the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire (Garcia-Chevesich and others 2004). Erosive forces were intensified on sites where the wildfire destroyed the vegetation that protected a site from the impacts of raindrops and splash, consumed the protective litter accumulations and other organic matter on the soil surface, and caused the formation of water-repellent soils. In particular, elevated post-fire soil erosion is more often found on hydrophobic soils than on non-hydrophobic soils (DeBano and others 1998; Neary and others 2005). The resulting increased soil erosion gradually declined as a partial cover of protective vegetation was re-established and the water repellency of the soils had declined. The cycle of increasing and then decreasing soil erosion on the Stermer Ridge watersheds after the Rodeo-Chediski Wildfire is similar to results found following other wildfires (Brooks and others 2003; DeBano and others 1998; Neary and others 2005).

Patterns of soil deposition following the wildfire were less predictable. Post-fire depositions of soil were generally dependent on the cumulative effects of the severity of the fire; the magnitudes of post-fire soil erosion; the hillslope topography and obstructions to the movement of soil particles; and the magnitudes, intensities, and timing of precipitation events after the fire and consequent overland flows of water. While some of these contributing factors were measurable in this study, others were not, therefore, resulting in the unpredictability of where soil depositions occurred.

Measurements of soil erosion and deposition are summarized by bar graphs that illustrate the magnitudes of the two processes for the sampling periods (figs. 17 and 18). Inferences relating to differences in and among the magnitudes of the two soil movement processes illustrated in the bar graphs are not necessarily valid, however, because frequency distributions of the data sets used in developing the bar graphs were mostly non-normal. Occurrences of significant differences in the measurements of soil erosion and deposition were determined through interpretations of the non-parametric tests selected for this purpose.

Soil Erosion

The largest magnitudes of post-fire soil erosion were measured on both of the Stermer Ridge watersheds in the first years following the Rodeo-Chediski Wildfire. The erosion rates then declined to the end of the study. No measurable soil erosion was observed on either of the watersheds in the fall of 2005 or in the following spring (fig. 17). It was possible that compensating amounts of soil erosion and deposition occurred on a watershed basis at those times. It was less likely that neither of the two processes occurred.

Soil erosion in the fall of 2002, as indicated by the measurements of soil pedestals shortly after the wildfire, was larger on Watershed A than on Watershed B (fig. 17). Similar differences in soil erosion on the two watersheds, as determined by the subsequent measurements of the capped erosion pins, continued through the spring of 2004. The differences in soil erosion on the two watersheds in the early years following the wildfire were attributed largely to the combined effects of the greater loss of vegetative cover, the more extensive formation of water-repellent soils, and the higher amounts of overland flow of storm runoff on Watershed A. Soil erosion on the two watersheds was similar from the fall of 2004 to the end of the study.

Comparisons of soil erosion measurements in the spring with those in the fall were made only in 2003 and 2004, when measurable soil erosion occurred in both of the seasons on both of the watersheds. While the seasonal measurements of soil erosion on Watershed A were similar in 2003, the measurements of soil erosion in the spring of 2004 were greater than those in the fall of that year. There were no differences in the seasonal measurements of soil erosion on Watershed B in either of these two years.

Measurements of soil erosion in the spring on Watershed A differed throughout the study period. However, the differences were inconsistent and not

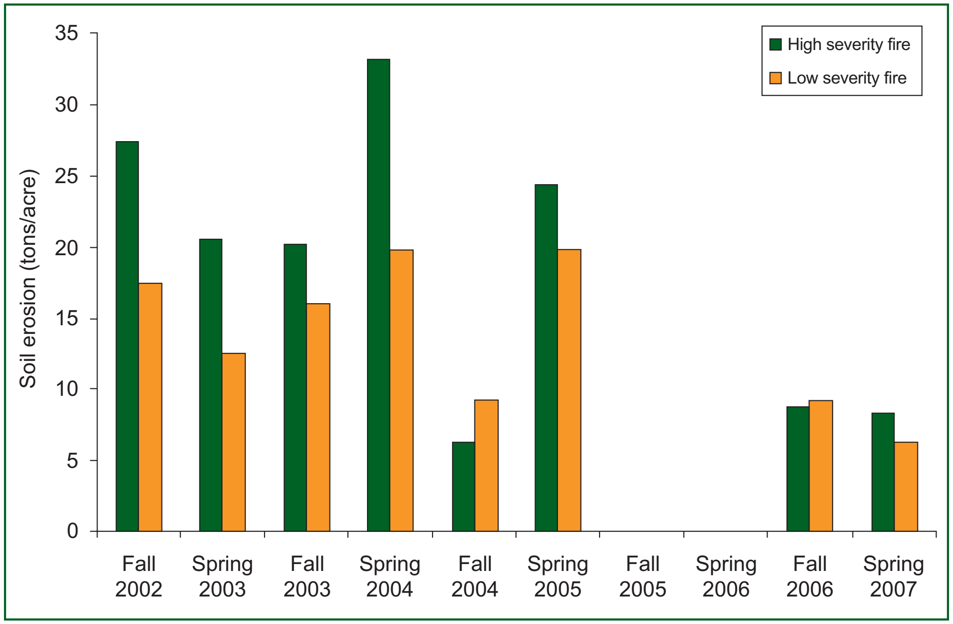


Figure 17—Soil erosion rates (tons/acre) on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire. Soil erosion in the fall of 2002 was estimated from measurements of soil pedestals. Otherwise, a sample of 12 or more plots was the minimum number of measurements necessary in analyzing these data sets.

related to the time since the wildfire occurred or the amount of seasonal precipitation prior to the measurements of soil erosion. Soil erosion measured in the fall on this watershed was not different. The measurements of soil erosion in the spring and in the fall were not significantly different on Watershed B.

Individual plots with measurable soil erosion were variable in number and spatial distribution throughout the study period. Comparatively, however, higher magnitudes of soil erosion tended to be found on sites located on steeper hillslopes with minimal post-fire vegetation than elsewhere on the watersheds.

Soil Deposition

There were no significant differences in the amounts of soil deposition on the two watersheds throughout most of the study period. It should be noted, however, that measurable soil depositions were not recorded on either of the watersheds in the spring of 2003, Watershed A in the spring of 2006, or Watershed B in either of the seasons in 2005 and the spring of 2006 (fig. 18). That depositions of soil were not measured on a watershed basis at these times could have been because equal magnitudes of soil erosion and soil deposition occurred. It was less likely that neither soil erosion nor soil deposition occurred on the hillslopes of the watersheds in the measurement intervals.

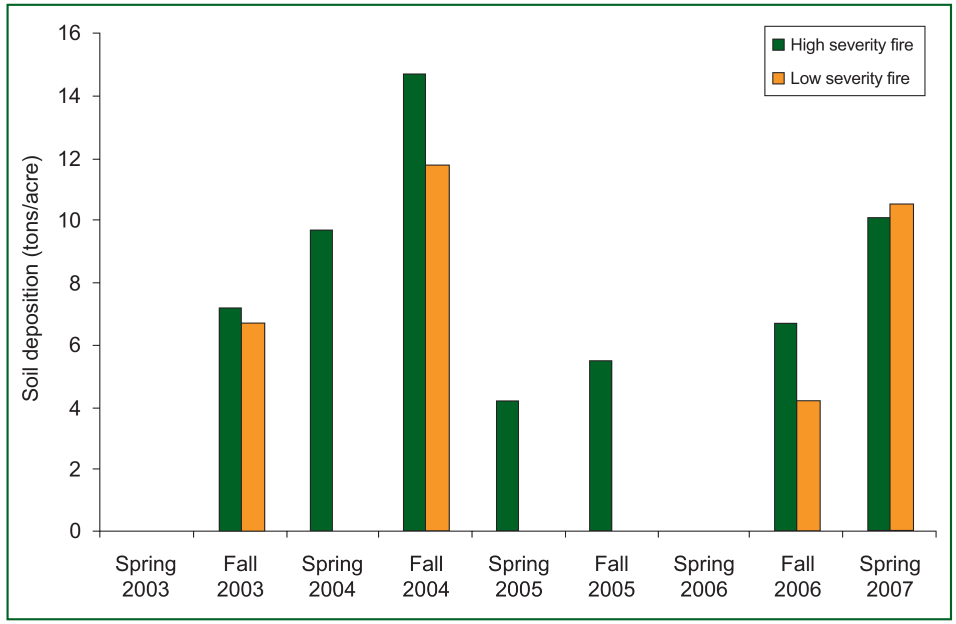


Figure 18—Soil deposition (tons/acre) on the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire. A sample of four or more plots was the minimum number of measurements necessary in analyzing these data sets.

Comparisons of soil deposition measurements in the spring with those in the fall were made in 2004 and 2005 on Watershed A—the only times in the study that measurable soil deposition were recorded on the watershed in the two seasons. In both of those years, however, there were no significant differences in the magnitudes of soil deposition (fig. 18). The number of plots was too small to adequately compare the depositions of soil on Watershed B in the two seasons in 2004 and 2005.

Differences in the amounts of soil depositions measured in the spring and in the fall were insignificant on the two watersheds throughout the study period. Neither time since the wildfire nor the amount of seasonal precipitation that occurred prior to the measurements of soil deposition were correlated with depositions of soil on the watersheds.

Actual plot locations where soil deposition was measured in either the spring or the fall were scattered on both watersheds, with little relationship to hillslope position or where slope gradients tended to decrease near to the stream channels of the watersheds. Neither the amounts nor the locations of soil deposition were related to the amounts or locations of where soil erosion was measured in the same time interval and vice versa. That is, soil deposition was less than soil erosion in some instances, while soil deposition was greater than soil erosion at other times. Therefore, it was not possible to trace the movement of eroded soil particles from one plot to its deposition on another plot or into a stream channel. The sampling procedures in this study were not designed for this purpose.

Other Studies on Impacts of Wildfire on Soil Movement

Earlier studies have also reported increases in soil erosion rates of varying magnitudes following a wildfire (Brooks and others 2003; DeBano and others 1998; Neary and others 2005; and others). In most cases, the increases in soil erosion were greatest on fires that were characterized by large percentages of high severity fire and where high-intensity, short-duration rains fell on mineral soil surfaces exposed by the wildfire. Most studies have often found that the post-fire rates of soil erosion started to decline toward pre-fire rates as a protective vegetative cover was re-established on the burned sites, and, as a consequence, the sites begin to recover from the impacts of the burn. A variety of fire-effects models are available to simulate these cycles of increasing and then decreasing post-fire soil erosion rates (Ryan and Elliot 2005).

Rich (1962) reported that depositions of soil following a wildfire were found mostly on sites where slope gradients decreased near stream channels, resulting in decreased velocities of overland flows of water. However, the measurements of soil deposition in that study were obtained on a watershed with steeper slopes and greater annual precipitation amounts than the Stermer Ridge watersheds. Erosive forces on the watershed studied by Rich (1962) were likely more intensive as a consequence.

Figure 19—The peak stormflow shortly after the wildfire, estimated on Stermer Ridge Watershed A, was nearly 2230 times that measured on the watershed before the fire. The 3-ft H-flume provides an approximate scale of the respective depths of the pre-fire and post-fire flows. (*Photo by Daniel G. Neary*.)

Peak Stormflows

The cumulative effects of losses of vegetative covers, decreases in litter and duff layers, formation of water-repellent soils, and other disruptions in the hydrologic functioning of the Stermer Ridge watersheds following the Rodeo-Chediski Wildfire were reflected to a large extent by the high peak stormflows that originated on the watersheds following the fire. Before precise stormflow measurements could be obtained following re-instrumentation of the watersheds with water-level recorders, historically high peak stormflows were estimated from high-water marks observed within, and occasionally above, the control sections. A peak of a stormflow event on Watershed A shortly after cessation of the burn was about 9.3 ft^3/sec or nearly 900 times what was measured before the wildfire (Ffolliott and Neary 2003; Neary and others 2005). The peak stormflow estimated on Watershed B at the same time was about 50 percent less, but it was still far in excess of pre-fire measurements.

A subsequent and significantly higher peak stormflow that originated on Watershed A within a week of the earlier stormflow was estimated to be 232 ft^3/sec or about 2230 times what measured before the burn (fig. 19). Though high peak flows are often observed following wildfire on smaller watersheds at the headwaters of larger river basins such as the Stermer Ridge watersheds, this latter estimate represented the highest known post-fire increase in peak stormflows in southwestern ponderosa pine forests (Ice and others 2004; Neary and others 2005). However, this estimated peak streamflow discharge was at the lower end of the ranges of streamflow discharges in the western United States, reported by Biggio and Cannon (2001) as cited by Neary and others (2005).

Comparatively high peak stormflows of varying magnitudes continued throughout the following summers of the study. However, the peaks gradually declined toward the pre-fire levels as hydrologic functioning of the two watersheds began to recover from the burn. This decline was particularly evident and more rapid on Watershed B, where the hydrologic functioning had not been severely disrupted by the burn.

Patterns of initially high and then declining peak stormflows following a wildfire have been reported elsewhere in southwestern ponderosa pine forests (Campbell and others 1977; DeBano and others 1998; Neary and Gottfried 2002; Neary and others 2005; and others). These extreme events accompanied by the transport of increased sediment have often led to significant changes in hydrologic functioning of the impacted stream systems. For the most part, peak stormflows are greatest the first year following fire and then decline thereafter and as watersheds recover. However, there is always the risk of higher peakflows in the second and subsequent years if unusual rainfall occurs. Steep terrain can also aggrevate this general statement about peak stormflow occurrence.

Post-fire stormflows with excessively high peaks are often associated with downstream flooding (Brooks and others 2003; Chang 2006; Neary 2002). Such a possibility was a concern to the people living downstream of the area burned by the Rodeo-Chediski Wildfire. In this regard, a high-intensity storm on the

White Mountains-Fort Apache Reservation dropped over 1.5 inches of rain onto a severely burned landscape on August 5, 2002—enough rainfall for meteorologists to call the storm a "10-year event." The increase in streamflow as a result of this storm registered a peak discharge of nearly 1200 ft^3/sec at a stream gauge on the Reservation maintained by the U.S. Geological Survey. This put the peak stormflow in the category of a "100-year event" at the site, which was one order of magnitude larger than expected (Ffolliott and Neary 2003). Fortunately, excessive downstream flooding did not occur as a result of this or other post-fire stormflow events originating on the burned area. However, this is a major problem for flood forecasters and emergency managers. In this case, a 10-year rainfall produced a 100-year streamflow. Data from other fires (Neary and others 2005) and the Rodeo-Chediski Wildfire have shown that post-fire flows can be 7, 50, 100, and, in this instance, 2230 times pre-fire flows for a given rainfall. Floods from post-fire rainfall events are likely to be way outside the natural range of variability and to produce serious post-fire effects. This situation can be aggravated if steep terrain is involved in the wildfire.

Water Quality Characteristics

Analysis of the sediment-laden water running off the larger areas burned by the Rodeo-Chediski Wildfire that was sampled downstream of the Stermer Ridge watersheds showed that the stormflows generated by the initial summer rainstorms following the burn contained large concentrations of organic debris, dissolved nutrients (including nitrogen, phosphorus, and carbon), and other chemical constituents that were released by the burning event (Ffolliott and Neary 2003). The elevated concentrations of nitrogen and phosphorus were attributed largely to the decomposition of the fire retardants that were dropped from the air to slow the advancement of the wildfire during the fire-suppression activities. Some of the increases in nutrients might have also originated from the ash and other plant residues that resulted from the wildfire.

Some investigators have reported little impact of fire in southwestern ponderosa pine forests on water quality constituents (DeBano and others 1998; Gottfried and DeBano 1990), while other researchers have found higher post-fire concentrations of chemical constituents such as calcium, magnesium, potassium, and combined organic-inorganic nitrogen in stormflow events shortly after a wildfire (Campbell and others 1977). However, the elevated post-fire concentrations often returned to pre-fire levels after the initial stormflow events. Such was observed in stormflows following the Rodeo-Chediski Wildfire.

The seemingly poor quality of water flows entering the downstream Roosevelt Reservoir shortly after cessation of the Rodeo-Chediski Wildfire was another possible concern (Neary and others 2005). The reservoir is the primary source of water for Phoenix, the largest city in Arizona, and the surrounding metropolitan communities. What made this concern more serious than might otherwise be expected was the fact that the body of water in the Reservoir at the time of the wildfire was less than 15 percent of its storage capacity. It was thought, therefore, that the elevated concentrations of sediments, organic debris, and nutrients in the water entering the Reservoir might threaten the aquatic life, leaving the reservoir water "lifeless" for months (Ffolliott and Neary 2003). However, that dire situation failed to materialize. Some fish died upstream and a few carcasses showed up at the diversion dam above the Reservoir, but the water in the Reservoir suffered little long-term environmental damage. Moreover, the anticipated formation of algae blooms that were predicted to consume much of the oxygen in the water did not occur. The debris in the flows of water that followed the wildfire was never a health risk for people as most of the pollutants were chemicals that were easily removed in downstream water-treatment facilities.

Impacts on Flammable Fuels

Pre-Fire Fuel Loadings

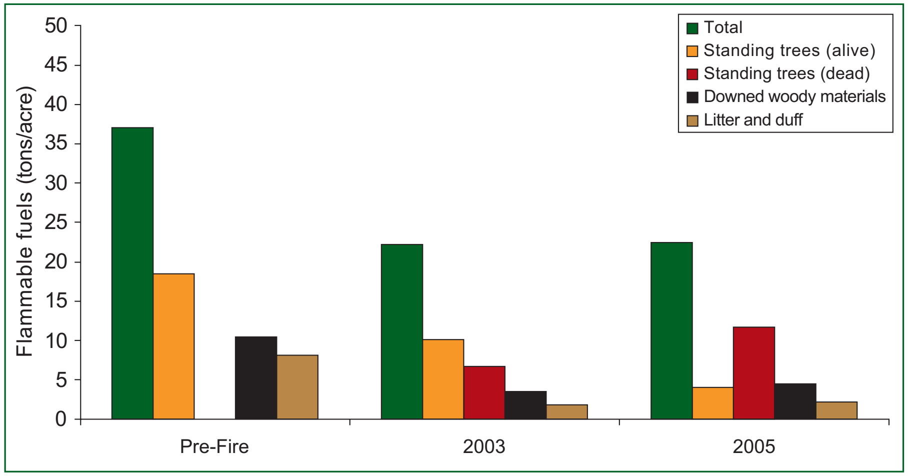
Standing trees were the largest fraction of the pre-fire flammable fuels on both of the Stermer Ridge watersheds (fig. 20). The comparatively smaller loading of standing trees on Watershed A was due to the smaller pre-fire volume of overstory trees (Ffolliott and others 2008). The estimated volume of pre-fire trees on this watershed was 1414 ft^3/acre in comparison to a volume of 1950 ft^3/acre on Watershed B. However, the estimated loadings of standing trees on both of the watersheds must be interpreted within the context of the volume table that was used to estimate the cubic-foot volumes of trees. This volume table (Ffolliott and others 1971) excluded the fuels contained in tree-tops, branches, and twigs and, therefore, underestimated the total fuel loading of the trees.

Post-Fire Fuel Loadings

One Year after Wildfire

Loadings of the flammable fuels one year after the burn on Watershed A provide an estimate of the initial impacts of the Rodeo-Chediski Wildfire on these fuels. Consumption of 40 percent of the pre-fire fuels on Watershed A was estimated to have occurred, mainly in the fraction of standing trees (fig. 20A). Almost 40 percent of the standing trees on this watershed died in the first post-fire year (Ffolliott and others 2008), lowering the estimated loading of this fraction to an unknown level. Charred (burned) wood that was contained in the trees reduced the volume of fuel that might otherwise have been classified as flammable. The estimated loading of this fraction, therefore, was probably overestimated. The extent that the overestimates and

[A]



[B]

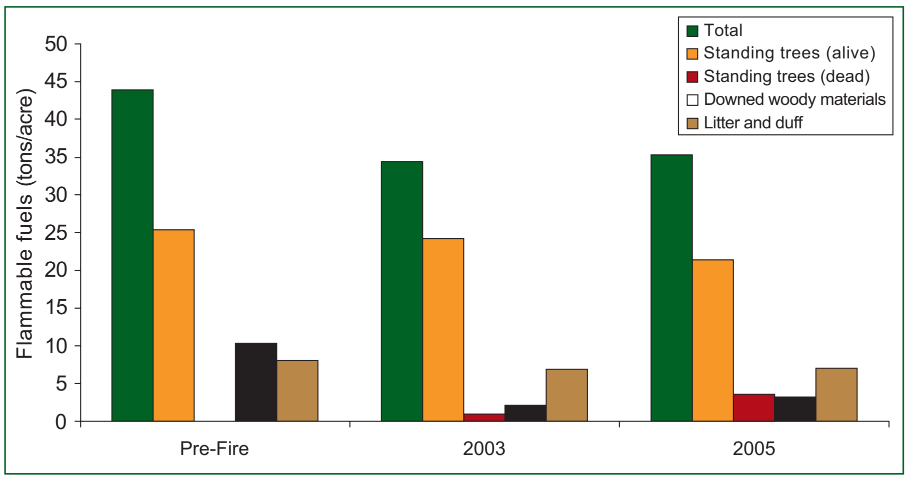


Figure 20—Average loadings of flammable fuel fractions before and after the Rodeo-Chediski Wildfire on the Stermer Ridge watersheds that were burned by (A) a high severity fire and (B) a low severity fire. Fine fuels consisting of herbaceous plants and shrubs are not plotted in this figure because the average loadings of this portion of the fraction were less than 0.1 tons/acre. Confidence intervals are not presented in this figure because of the variable procedures applied in obtaining the source data.

underestimates of the standing trees fraction attributed to the volume table used in the study were compensating is unknown. Nevertheless, the large loss of fuels in this fraction was expected because most of the pre-fire trees in the reconstructed overstory on this watershed had been either killed by the fire or suffered severe damage to their crowns (Ffolliott and others 2008).

About 20 percent of the pre-fire loadings of the fuels on Watershed B was lost and, therefore, these fuels were presumed to have been consumed in the wildfire, according to estimates made one year following the burn (fig. 20B). This result was also expected, as the portions of Watershed B that burned were impacted mostly by surface fire. As a consequence, less than 5 percent of the pre-fire trees on this watershed were killed by the wildfire (Ffolliott and others 2008).

One year after the burn, loading of downed woody materials was greater on Watershed A than on Watershed B (fig. 20). This result was attributed largely to the fact that trees that were killed or severely damaged on Watershed A started to fall to the ground and, therefore, were included in the estimate of this fuel fraction. Otherwise, the accumulations of stem sections, branches, and twigs on the forest floor were largely the same on both of the Stermer Ridge watersheds.

Loading of litter and duff on Watershed B was almost 3.5 times greater than the loading of this fraction on the other high severity fire watershed (fig. 20). While relatively small in terms of the loadings of all of the fuels, this difference was a consequence of the increase in litter fall from the standing trees on Watershed B that had survived the burn but with slightly scorched crowns. Because most of the trees on Watershed A were killed outright by the wildfire or died within two years (Ffolliott and others 2008), post-fire litter fall was inconsequential. A similar result was reported by Davis and others (1968), who compared the litter fall from the trees with crowns that were "slightly damaged" by a fire in a southwestern ponderosa pine forest to that from trees on unburned sites.

Loadings of fuels that consisted of herbaceous plants and shrubs were less than 0.1 tons/acre on the two watersheds in the first post-fire year. Differences between the two watersheds had little significance.

Three Years after Wildfire

Changes in the loadings of flammable fuels on the Stermer Ridge watersheds three years following the wildfire in relation to the loadings of the fuels one year after the burn were comparatively small, with the exception of the standing trees (fig. 20). Nearly 75 percent of the fire-damaged trees on Watershed A had died by the third post-fire year. This increase in mortality translated into an increase of 15 percent in the loading of standing dead trees over that observed one year after the wildfire. Dead trees on the ground were included in the estimate of downed woody materials when occurred on the measurement plots.

The continuous falling of dead and dying trees and needles from crown-scorched trees to the ground was the main factor that contributed to the small increases in downed woody materials and litter and duff layers (fig. 20). The increase in the fine fuels represented by the herbaceous plants and shrubs was inconsequential.

Following Salvage Cutting and Fuel Reduction Treatments

The salvage cutting and fuel reduction treatments on Watershed A also impacted the post-fire loadings of fuels in that the fraction that consisted of standing trees had been largely removed from this watershed. However, the loading of stem sections, branches, and twigs on the forest floor increased significantly (fig. 21). This fraction was about 2.5 times (or about 7.4 tons/acre) greater following completion of the salvage cutting and fuel reduction treatments than was estimated three years following the wildfire. Almost 90 percent of the increase in this fraction was contained in materials 1 inch and larger in dbh that were mostly the residue of the treatments. Therefore, while the likelihood of a crown fire occurring on this watershed in the foreseeable future has been eliminated, the possibility of surface fire remains.

Other Studies on Impacts of Wildfire on Flammable Fuels

This analysis of the Rodeo-Chediski Fire's impacts on the loadings of flammable fuels on the Stermer Ridge watersheds should be interpreted only as a case study of the impacts on the fractions studied. The extent to which these findings might apply elsewhere in the Southwestern Region is unknown. However, the impacts of wildfire on the loadings of flammable fuels in southwestern ponderosa pine forests and how combustion of these fuels has shaped the post-fire structure of the forests have been considered in a general framework in other studies (Arno 2000; Cooper 1961; Daubenmire 1943; Martin and others 1979; Weaver 1951). Zimmerman (2003) presented a comprehensive overview of this topic in a paper on fuels and fire behavior in southwestern ponderosa pine forests.

Figure 21—Accumulations of stem sections, branches, and twigs on the forest floor following salvage cutting and fuel reduction treatments on Stermer Ridge Watershed B that was burned by a high severity fire. The remaining standing trees are mostly crown-killed Gambel oak (*Quercus gambelii*) with little merchantable value. The piles of logs and larger trees in the background were converted into chips to generate electricity for local towns. (*Photo by Peter F. Ffolliott.*)

Summary

Impacts on Ecosystem Resources

The Rodeo-Chediski Wildfire altered the pre-fire stand structures of tree overstories on the Stermer Ridge watersheds, with the impacts more drastic on Watershed A due to the high severity fire than on Watershed B that was burned by a low severity fire. The pole class of trees suffered higher mortality than trees in the sawtimber classes. Most of the trees that survived the wildfire but were severely damaged by the burn died within two years of the burn and had started falling to the ground. Salvage cutting and fuel reduction treatments on Watershed A eliminated most of the remaining standing trees. Stocking of ponderosa pine seedlings after the wildfire was insufficient on either of the watersheds to sustain a ponderosa pine forest.

Production of herbaceous plants and grass and forb components of the herbaceous plants were generally greater throughout the study on Watershed A. Larger reductions of the tree overstory and more complete removal of the litter and duff layers on the watershed contributed to this finding. Mullein represented almost one-half of the production of the forbs on the severely burned watershed at its peak occurrence, but this early-seral species contributed less to the lower level of forb production on the watershed that was burned by a low severity fire. Mullein declined in occurrence after dominating the forb component of herbage on both of the watersheds in the early and middle years after the wildfire.

Impacts of the wildfire on the indigenous ungulates and small mammals on the Stermer Ridge watersheds differed in both magnitude and duration following the burn. Increased presence of elk in the third year after the burn on Watershed A was attributed to the increase in preferred forage plants. The availability of protective cover was a secondary factor of importance. The smaller quantities of browse preferred by mule deer resulted in lower occurrences of this ungulate relative to that of elk on both of the watersheds. The limited presence of cottontail on both of the watersheds was probably the result of the more limited availability of preferred forage plants. The apparent departure of Abert's squirrel from Watershed A was caused largely by the loss of preferred feed trees in the high severity fire. However, the squirrel's feeding activities by were observed two years after the wildfire on Watershed B, indicating their return to this watershed. This finding was attributed to the fact that most of the squirrel's preferred feeding trees survived the fire.

Not knowing the structure of bird communities on the Stermer Ridge watersheds before the wildfire precluded comparisons of pre- and post-fire species and numbers. Nevertheless, the numbers of birds in some guilds declined, while birds in other guilds were affected less by the wildfire. Because of varying responses of birds to the Rodeo-Chediski Wildfire, a mosaic of habitat conditions on a scale larger than that on the Stermer Ridge watersheds seems necessary to sustain a full suite of bird species following the burn.

Impacts on Hydrologic Functioning

Nearly two-thirds of the soils on Watershed A exhibited strong water repellency at the surface immediately after the fire, while moderate water repellency was found on one-third of the soils. Strong water repellency was measured at the soil surface on approximately one-third of Watershed B, with 15 percent of the soils having moderate water repellency and almost 50 percent of the plots with slight or no water repellency (fig. 16B).

Most of the water-repellent layers in soils dissipated on both of the Stermer Ridge watersheds within three years of the wildfire. The decline in the occurrence of water-repellent soils on the two watersheds was attributed to several processes. Downward erosion of the soil into and through the hydrophobic layer and repeated exposures of the soils to cycles of wetting-drying and freezing-thawing events also led to a decrease in the degree of water repellency in the soils. Differences in the occurrences of water repellency on the two watersheds, the respective levels of the water repellency, and the patterns of decline in the water-repellent soils between the spring and fall seasons were inconsequential.

One impact of the wildfire on the hydrologic functioning of the Stermer Ridge watersheds was a decrease in the interception of precipitation due to the loss of tree overstories and consumption of litter and duff layers by the burn. Re-establishment of trees on the high severity fire areas is unlikely to occur without artificial regeneration. Litter and then duff layers will not re-form in the absence of trees.

Transpiration was another component of the water budget that was altered by the Rodeo-Chediski Fire. While sap flow was detected in all of the limited measurement trees, the few repeated measurements (two measurement for each tree sampled) were insufficient to differentiate with confidence the transpiration losses on the two watersheds. Nevertheless, it was concluded that the loss of water by the transpiration processes was reduced on both of the watersheds, with comparatively higher losses on Watershed A because of the greater mortality of trees.

Infiltration of water into the soil was not measured in the study. It was assumed, however, that formation of water-repellent soils in the early years of the study inhibited the infiltration rates on the watersheds, causing much of the precipitation that reached the soil surface to increase in surface runoff flows. This phenomenon was more pronounced on Watershed A because of the more extensive formation of strong water repellency associated with high severity fire. Water-repellent soils, and hence reduced infiltration rates, persisted up to three years on the watersheds, after which infiltration rates probably increased as water repellency declined.

Destruction of the vegetative covers and litter and duff layers coupled with formation of the water- repellent soils on the Stermer Ridge watersheds resulted in peak stormflows of historical magnitudes following the wildfire. The highest of the peak stormflows were observed following the high-intensity, short-duration rainfall events that occurred shortly after cessation of the wildfire. While post-fire stormflows of exceedingly high peaks are often characteristic of flooding regimes, downstream flooding following the Rodeo-Chediski Wildfire did not occur. Quality characteristics of the water flowing from the Stermer Ridge watersheds were not measured in this study. However, it was ultimately determined that the concentrations of sediment, nutrients, and other debris in the water flowing from the area that was burned by the wildfire presented few environmental problems.

Impacts on Flammable Fuels

Estimates made one year post-fire indicated that 40 percent of the pre-fire fuels on Watershed A had been eliminated, mostly in the standing trees fraction of the total watershed fuels. Almost 40 percent of the remaining standing trees were dead by that time, reducing the loading of the fraction. Nearly 75 percent of the pre-fire trees had died by 2005, three years after the wildfire, with this increase in mortality translating into an additional increase of 15 percent in loadings of standing dead trees in relation to that one year after the fire.

Estimates made one year after the wildfire indicated that about 20 percent of the pre-fire loadings of flammable fuels on Watershed B had been consumed by the wildfire. Less than five percent of the pre-fire trees had been killed. This finding was not surprising, however, as much of the watershed was impacted only by low severity surface fire. The greater loading of litter and duff on the watershed at this time was attributed to the litter fall from standing trees that survived the burn with slightly scorched crowns. Post-fire litter fall on Watershed A was minimal because most of the trees were killed or severely damaged by the wildfire and died within two years.

Salvage cutting and fuel reduction treatments implemented on Watershed A impacted the post-fire loadings of flammable fuels. The fraction consisting of standing trees was essentially eliminated on this watershed. However, loadings of stem sections, branches, and twigs on the forest floor increased significantly as a result of the treatments. Therefore, while the probability of future crown fire has been reduced, the possibility of surface fire on the watershed remains.

Management Implications

Recovery of the Stermer Ridge watersheds from the Rodeo-Chediski Wildfire has been related to the respective fire severities that the two watersheds experienced. Watershed A that was burned by a high severity fire underwent a conversion from a southwestern ponderosa pine forest ecosystem to a landscape of grasses, forbs, and a few shrubs. This post-fire cover of herbaceous plants and shrubs is likely to persist in varying forms into the future in the absence of artificial regeneration of the tree species. Recovery of the hydrologic functioning on this watershed has begun on a limited scale with the elimination of fire-induced water-repellent soils and the resulting decrease in the magnitudes of overland flows of water; the reduction in hillslope soil erosion and deposition and, therefore, sediment production; and the gradual but continuous return of peak stormflows to pre-fire levels. It is anticipated, however, that the overall hydrologic functioning of this watershed will not approach pre-fire conditions for many years. Flammable fuels represented by standing trees have been eliminated on the watershed, but there has been an increase in stem sections, branches, and twigs on the forest floor. Therefore, while the possibility of a future crown fire has declined, the potential for surface fire remains.

Much of Watershed B was either burned at a low severity or unburned. The sites that were burned are slowly recovering from the impacts of the wildfire, a progression that is likely to continue into the future in the absence of wildfire or other major disturbances. Relatively small areas of Watershed B were burned at higher fire severities, but those sites were isolated and scattered throughout the landscape. Much of the hydrologic functioning of this watershed is returning slowly to its pre-fire level. The post-fire loadings of flammable fuels were largely unchanged from their pre-fire estimates. Consequently, this watershed remains vulnerable to future wildfire events.

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