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INTRODUCTION

The Spring meeting of the Hydrology Section of the Arizona-Nevada Academy of Science took place in Flagstaff, Arizona on April 17, 1999. We want to extend our thanks to Ms. Louella Holter, Bilby Research Center, Northern Arizona University, for her editorial assistance on this publication.

Malchus B. Baker, Jr.

WATER-REPELLENT SOILS: WHAT THEY ARE AND WHY THEY ARE IMPORTANT

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Awareness of water-repellent soils began with an observation made during a soil organic matter investigation at the beginning of this century. There was mainly academic interest in water-repellent soils until the 1960s, at which time it was realized that this unique soil property had far-reaching implications in the management of agricultural and wildland ecosystems. Since the 1960s, interest in these soils has expanded both geographically and scientifically. The importance of water-repellent soils was the theme for a conference held in September 1998 in Wageningen, the Netherlands. Speakers at this conference came from throughout the world to report on a wide range of topics related to soil water repellency. Specific topics included the origins of water repellency, assessment of water repellency, occurrences and consequences of water repellency, water flow and transport in water-repellent soils, and the amelioration of water-repellent soils. This paper reviews the evolution of the interest in these soils and some of the major findings reported at the Netherlands conference.

Normally, dry soils have an affinity for adsorbing liquid and vapor water. This occurs because there is strong attraction between the mineral soil particles and water. Not all soils, however, display these wettable characteristics; some repel water instead. For example, when water droplets are placed on the surface of an air-dry soil that is water repellent, the droplets bead up. Water will not penetrate because the mineral soil particles are coated with hydrophobic substances that repel water. The degree of water repellency is related to the contact angle between the water droplet and the water-repellent soil surface. Also, a so-called effective contact angle has been calculated for soils using a version of the capillary rise equation.

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Chemically, hydrophobic substances have no polarity and therefore do not attract water. This resistance to wetting affects water movement into and through soils, as will be discussed in detail in a second paper on this issue (DeBano 1999). The discussion here, however, is limited to the general nature of soil water repellency, the historical activities and attention focused on water repellency, the problems arising from this unique soil condition, and the various remedial treatments used to ameliorate water repellency. Other topics that captured interest at a recent international workshop on water repellency in soils are also discussed.

The Nature of Water Repellency

The cause of water repellency in soils is related to organic matter. Water repellency can result from the following processes involving organic matter: (a) an irreversible drying of the organic matter (e.g. the difficulty encountered when rewetting dried peat materials); (b) production of leachates, which coat mineral soil particles (e.g. coarse-grained materials easily made water repellent by plant leachates); (c) coating of soil particles with hydrophobic microbial byproducts (e.g. fungal mycelium); (d) intermixing dry mineral soil particles and dry organic matter; and (e) vaporization of organic matter and condensation of hydrophobic substances on mineral soil particles during fire (e. g. heat-induced water repellency).

Attempts have been made to characterize water repellency chemically. The chemistry, however, is complex and elusive and it appears that numerous organic compounds are responsible for producing water repellency. These compounds can be either synthesized in nonfire environments by microorganisms or formed during soil heating by fires. Many chemicals have been proposed as sources of water repellency: a mixture of aliphatic hydrocarbons, aromatic compounds, esters, polysaccharides, humic and fulvic acids, fatty acids, and polymethylenes (Wallis and Horne 1992). The

severity of water repellency depends not only on the organic matter, but also on soil texture, soil water content, and fire severity.

Historical Interest

Water repellency has been a concern of both scientists and land managers for well over a century. During this time, the interest in water repellency has evolved from an isolated scientific curiosity to an established field of science that is recognized worldwide.

Interest in water-repellency phenomena began during the nineteenth century, when it was reported as being associated with organic matter and its decomposition, particularly by fungi. Reports on fungal decomposition of organic matter by Waring in 1837 (Bayliss 1911) were probably the first publications describing the effect of mycelium growth on water absorption by soil. Wetting resistance was associated with a phenomenon known as fairy rings. The term "fairy ring" was used by these early investigators to describe the arrangement of plants (usually grass or crop plants) in an approximate circular form where plant growth on the inside of the circle was stimulated. This inner circle of healthy plants was surrounded by circles of bare ground or concentric zones of withered plants. In many cases fairy ring phenomena were so abundant that they materially affected the yield of grasslands and agricultural crops.

The interest in organic matter, and specifically humic substances, continued into the earlier part of the twentieth century. In 1908, Schreiner and Shorey (1910) reported studying a soil "that could not be wetted, either by man, by rain, irrigation or movement of water from the subsoil." Shortly thereafter it was reported that in eastern Colorado, grassland soils infected with fleshy fungus (*Agaricus tabularis*) had lower soil water contents than bare areas and the inner and outer vegetated rings, particularly in the surface soil early in the growing season (Shantz and Piemeisel 1917). Papers describing water-repellent soils began appearing in scientific journals during the 1940s and 1950s. Studies in Florida reported wetting resistance in citrus orchards (Jamison 1947). Elsewhere in the world, water repellency was reported in New Zealand and Germany.

The 1960s witnessed the beginning of national and international interest in soil water repellency. Several noteworthy activities occurred during this decade. First, a large number of papers were published on water repellency, primarily by scientists in Australia and the United States. For example, between 1960 and 1970, more than 90 publications dealing with various aspects of water repellency were published. A second significant landmark was reflected in two publications: one describing the measurement of liquid-solid contact angles in soil and sand (Letey et al. 1962a) and a second describing the influence of water-solid contact angles on water movement in soil (Letey et al. 1962b). The third milestone was the summarization and synthesis of all available knowledge on water repellency at the first international conference on water repellency held in June 1969 at the University of California, Riverside (DeBano and Letey 1969). Also during the 1960s, excessive postfire erosion following wildfires was shown to be related to the water-repellency of soils formed during fires (Krammes and DeBano 1965). This discovery stimulated a search for a better understanding of the chemical properties of hydrophobic substances and the use of nonionic wetting agents for remedial treatments.

Following the 1968 conference, the interest in water repellency expanded both nationally and internationally. Not only were land managers in the United States interested in water repellency, but it was reported in Italy, Spain, Portugal, Greece, Poland, Taiwan, Egypt, India, New Zealand, Australia, South Africa, Chile, Germany, the Netherlands, and Belgium.

Associated with this expanding interest were the applications of emerging scientific knowledge in several fields, particularly in soil physics, where water movement into and through soils was being characterized as a "fingering" process. Understanding the fingering process depended upon having a knowledge of the physics of water movement through heterotrophic systems with unsteady saturated and unsaturated flow processes occurring simultaneously, heterogeneous horizontal layers (e.g. soil horizons or other subsurface strata), preferential and lateral flow pathways, and hysteresis.

Management and Amelioration

Water repellency in soil created management problems almost from its initial identification early in the twentieth century. The attention received by water repellency in soils, however, has evolved from what was previously an isolated academic curiosity in the early twentieth century to present-day worldwide concern among scientists and managers of both agricultural lands and wildlands. Approaches for ameliorating water repel-

lency also evolved concurrently as the effect of water repellency on water movement processes became better understood.

The first descriptions of difficult-to-wet soils were reported as incidental observations during an organic matter study in California (Schreiner and Shorey 1910) and a study of fungal infection of grasslands in Colorado (Shantz and Piemeisel 1917). However, specific examples of decreased productivity of citrus orchards were first reported in the 1940s (Jamison 1947), and the adverse effect (dry patch) of water repellency on the productivity and aesthetics of golf greens began attracting serious attention during the 1960s (Morgan et al. 1966). Also during the 1960s, water repellency was reported to reduce productivity of pastures in Australia (Bond 1964). Also, the role of water-repellent soils in erosion and postfire management of chaparral areas in California that had been burned by wildfires was attracting attention during the 1960s (DeBano et al. 1967).

Since the 1970s, the interest in water-repellent soils has continued to grow. It was estimated in the early 1990s that about 5 million hectares of land in Australia and New Zealand were adversely affected by water repellency, resulting in a financial loss of more than 150 million dollars. In the Netherlands, not only do water-repellent soils affect productivity, but they accelerate the movement of contaminants (e.g. fertilizers, pesticides) into the groundwater tables via preferential water flow. As a consequence, a widespread interest has developed in creating models that can be used to describe the transport of chemicals through soils under field conditions (Flüher et al. 1996). Soil classification systems are now being developed to reflect the susceptibility of soils to the rapid transport of water and associated chemicals into groundwater (Quisenberry et al. 1993).

Fire-induced water repellency remains a present-day concern following both wild and prescribed fires. Attempts are currently being made to develop infiltration models that will account for the effect of water-repellent soils. Water repellency also is being recognized as an important factor in erosional processes in nonfire environments, particularly in sand dunes along the west coast of the Netherlands (Witter et al. 1991). Concerns are being focused on the spatial variability of fire-induced water repellency (Doerr et al. 1998) and the relationship between the spatial distribution of water repellency and erosion potential produced during prescribed burning (Robichaud 1996).

Several other, less widespread concerns about water repellency have been reported during the last 30 years. For example, localized areas of highly water repellent soils are created by oil spills, requiring intensive remedial efforts (Roy and McGill 1998). Furthermore, water repellency is becoming recognized as a key mechanism responsible for the self-cleaning features of plant surfaces (Neinhuis and Barthlott 1997). The diminished aesthetics and playing qualities of golf greens that are afflicted with the age-old problem of "dry patch" continue to be a concern to golf green supervisors (Hudson et al. 1994).

An increased interest in water repellency has also been focused on wildland soils, in both fire and nonfire environments. Water-repellent soils have been reported in several wildland environments, including dry sclerophyll eucalyptus in Australia (Crockford et al. 1991), eucalyptus forests in South Africa (Scott 1991), eucalyptus and pine forests in Portugal (Doerr et al. 1996), and windbreaks in Taiwan (Lin et al. 1996). Also, the fire-induced water repellency described above continues to have its primary impact on soil-water relationships in wildland ecosystems.

The first remedial treatments used for water repellency were wetting agents (or surfactants). Considerable research was conducted by the scientists at the University of California, Riverside, on the effectiveness of specific surfactants for treating water repellency in soils. Nonionic wetting agents were rapidly identified as being chemicals that could readily wet-up highly water repellent soils (Letey et al. 1975; Miller and Wilkinson 1977). Water-repellent thatch material on golf courses was effectively treated with these wetting agents. Wetting agents were also tested as a treatment for ameliorating heat-induced water repellency in brushland soils burned during wildfires. Although anionic wetting agents effectively reduced erosion on small burned plots, an effort to use operational-level wetting agent treatments to reduce soil erosion on burned watersheds was not successful (Rice and Osborn 1970). Another technique used to ameliorate water repellency in dry-patch turf areas of Australia consisted of mixing loamy soil with naturally occurring water-repellent sands to improve their wettability (Bond 1978).

Coping with water-repellent soils continues to present a challenge to managers of agricultural and pasture lands worldwide into the 1990s (Capriel 1997). Interest has continued in the usefulness of wetting agents as a remedial treatment for water repellency (Effron et al. 1990), although this treatment does not capture as much interest as it

did during the 1960s and 1970s. Wetting agents have also been used to increase the effectiveness of irrigation (Wallis et al. 1990).

Remedial treatments other than wetting agents are being tested extensively at present in Australia and New Zealand (Blackwell et al. 1993). These treatments include direct drilling, wide furrow sowing, and the use of microorganisms and fertilizers to stimulate microbial breakdown of water repellency. Soil claying, a treatment that involves mixing large amounts of clay in the upper water-repellent layer, has received widespread use in Australia (Carter and Hetherington 1997). High pH soil treatments have also been used to improve the wettability of soils supporting golf greens (Karnok et al. 1993).

An International Workshop on Soil Water Repellency

During the week of 1 September 1998, a 3-day international workshop was held, titled "Soil Water Repellency: Origins, Assessment, Occurrence, Consequences, Modeling, and Amelioration" at the SC-DLO Winand Staring Centre for Land, Soil and Water Research, in Wageningen, the Netherlands. The conference was planned to be held 30 years after the first International Conference on Water Repellency at Riverside, California in 1968 (DeBano and Letey 1969). An important part of the 1998 workshop was to review the progress and changes in the field that had occurred during the past 30 years.

The increased interest in water repellency since the 1968 conference was demonstrated by the large number of scientists from many parts of the world who presented papers at the Wageningen workshop. The 1998 workshop was attended by about 70 scientists from 14 countries: Australia, Belgium, Canada, Denmark, Germany, Greece, the Netherlands, New Zealand, Portugal, Russia, South Africa, Spain, the United Kingdom, and the United States. In contrast, only scientists from Australia, England, and the United States participated in the earlier conference at Riverside.

The subject material at the two conferences was somewhat similar. In 1968, the major areas of interest were characterizing water repellency, movement of water through water-repellent soil systems, use of wetting agents as remedial treatments, fire-induced water repellency, hydrophobic compounds that produced water repellency, and the effect of water-repellent soils on plant growth. However, during the Wageningen workshop, entire sessions were devoted to origins and assessments of water repellency, occurrence and consequences of water repellency, flow and transport in water-repellent soils, and amelioration of water-repellent soils.

Significant advances in many areas were apparent during the past 30 years, though some age-old problems remain unsolved. For example, substantial progress in soil physics during the past 30 years has provided a much better understanding and characterization of water movement through water-repellent soil systems. Along with progress in scientific theory came the development of improved technology, now capable of more thoroughly analyzing the composition of organic matter and of measuring soil water changes continuously under field conditions. Considerable progress has been made in developing specific mechanical and agronomic techniques for ameliorating water-repellent soils, particularly in Australia and New Zealand. Chemical treatments, such as wetting agents, still remain important but are sharing the limelight with physical approaches (e.g. claying or furrowing). The processes involved in creating fire-induced water repellency are understood better now than 30 years ago, but effective postfire remedial treatments are still lacking. The century-old problem of fairy rings, dry patch, or thatch accumulation still remains a high priority in the management of golf greens. The most important impression obtained when reviewing the progress during the past 30 years is that water repellency in soils has evolved from a limited scientific curiosity to a full-fledged field of science that has attracted the efforts of scientists and managers worldwide.

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INFLUENCES OF MAP SCALE ON DRAINAGE NETWORK REPRESENTATION

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In hydrologic and geomorphic research, the stream channel network is the basis for modeling and interpreting surface processes. Although the automated extraction of channels from a digital elevation model is becoming more widely applied, the standard method for delineating streams has been to capture them from aerial photographs and topographic maps. Much of the original research in basin geomorphology and scaling issues in hydrology was based on network data derived from commonly available maps. A serious flaw is inherent in this approach, however; the number of streams, basin order, and the resulting network complexity is primarily a function of the scale of the base maps. Hydrologic detail is lost when small-scale maps or photographs serve as the basis for interpretation. This paper presents the results of a study using multiple maps at a variety of scales (1:12,000, two sets at 1:24,000, and 1:62,500) on the Walnut Gulch Experimental Watershed in southeastern Arizona. Significant differences in stream order, network complexity, channel length, and runoff efficiency were found as a function of scale. These differences have potentially significant implications for current research issues on the watershed, including hydrologic modeling efforts incorporating routing, watershed analyses, and multiscale watershed modeling.

The stream channel network often forms the basis for watershed, basin, and regional scale assessments of hydrologic and geologic systems. Watersheds may be characterized according to their drainage structure, which is determined by the underlying lithology and topography. Drainage patterns form in response to energy forces such that total channel length is minimized and drainage efficiency is maximized. A number of factors influence how a watershed conveys water and sediment. Rather than attempting to quantify each of these factors, it is convenient to use more readily measured geomorphic properties as proxies for other watershed characteristics. In this manner, inferences regarding hydrologic response and comparisons among catchments for classification and comparative analyses can be made (Gordon et al. 1992). This paper investigates the role that base maps have on such analyses in a semi-arid rangeland.

Intensive watershed characterization is time consuming and costly when done by traditional means. This cost has been an impediment to large-scale studies requiring morphometric measures. The advent of geographical information systems (GIS) allows for the rapid and precise determination of a host of channel and watershed characteristics. Caution is advised when undertaking these analyses. It is especially important that limitations imposed by the choice of base data be clearly defined and understood by the researcher.

Several factors are critical to the determination of stream channel location on the watershed. Most obvious is map or photo scale. At larger scales, objects are more readily detectable, and smaller features may be discriminated. In addition to the limitations on interpretation inherent in using small-scale maps, decreasing the scale also reduces the amount of information that can be included on a map for presentation. Hence, even though USGS 7.5' and 15' topographic maps may be based on the same photographic source, the small-scale 1:62,500 15' quadrangles will convey less information. Perhaps more relevant to photointerpretation is that image resolution (e.g. ground resolution distance) plays a key role in the discrimination of landscape features. Decreasing the pixel size within an image reveals smaller features, such as first-order channels or instream channel bars. Poor contrast or reduced image quality can also adversely affect photointerpretation.

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Modeling and interpretation of hydrologic processes is often reliant on a high-quality representation of the hillslope and drainage system within a watershed. Rainfall-runoff processes in the semi-arid Southwest are characterized as intermittent, high-intensity events during which water is rapidly transported through a watershed. Due to the low water tables and gravelly soils, high transmission losses predominate. Because transmission losses operate within the drainage system, adequate channel representation is important for accurate runoff modeling. Erosion and sediment transport are dominated by actions occurring within stream channels; thus, a sound depiction of the channel network is necessary for modeling and understanding the processes involved in sediment movement.

Channel morphometric measures have long been used as tools for watershed assessment and classification. The concept of channel ordering was first outlined by Horton (1932), and was later amended by Strahler (1952) to decrease the subjectivity of the analysis. The Horton-Strahler method denotes the uppermost channels, those with no incoming tributaries, as first order. Where two first-order channels converge, the downstream segment is ranked as second order. If two channels of different orders converge, the downstream segment is ranked as the higher of the two channels. In this manner, a watershed is characterized with an abundance of lower-order channels, with successively fewer higher-order channels. In his seminal research into the relationship between channel systems and their contributing areas, Horton (1945) introduced several laws governing watershed-channel relations as a function of scale, relying heavily on channel ordering. This research led to a cavalcade of geomorphologic and hydrologic research on the characteristics of drainage basins and their influences on hydrologic and erosional response (see Abrahams 1984 for a critical review).

Horton's law of stream numbers stipulates that the number of channels of a given order will decline according to a given ratio for a basin (eq. 1), whereas the law of stream lengths governs the increase in average channel length by increasing order (eq. 2), and the law of stream areas illustrates the increase in support area by order (eq. 3). As shown, these so-called laws of drainage network composition approximate to geometric progressions of inverse (law of stream numbers, eq. 1) or direct (laws of stream lengths, eq. 2, and stream areas, eq. 3) form:

$$R_B = \frac{N(w)}{N(w+1)} \quad (1)$$

$$F_L = \frac{L(w+1)}{L(w)} \quad (2)$$

$$R_A = \frac{A(w)}{A(w-1)} \quad (3)$$

where (w) denotes a given channel order, (w+1) denotes the next higher order, N is the number of channels, L the average channel length, A the watershed area, R_B the bifurcation ratio, R_L the length ratio, and R_A the area ratio. Various researchers have demonstrated that bifurcation ratios range between 2 and 5, with an average in the United States of approximately 3.5 (Leopold et al. 1964), underscoring the preponderance of lower-order channels on a landscape.

The influence of map scale on such Horton-style analyses is intuitive. With a decline in the number of lower-order channels resulting from a decrease in map scale, the overall ordering arrangement of a basin can be profoundly affected. Leopold et al. (1964) described an ordering analysis conducted in New Mexico using different map sources. With a 1:24,000 USGS topographic map, the basin was determined to be first order (a single channel), but more detailed maps showed the basin to be fifth order. Thus, ordering techniques are scale dependent and should be used with caution in comparative analyses or for watershed assessment.

Drainage density is often used in geomorphic analyses because it reflects the underlying geology, climate, soil, vegetation, and historical use of a region. Watersheds with high sediment yield, such as those found in semi-arid regions, generally have high drainage densities relative to their counterparts in more humid climates. Drainage density (R_D) is defined as the cumulative channel length (L) for n channels divided by the watershed area (A), as shown in equation 4.

$$R_D = \frac{\sum_{1}^{n} L}{A} \quad (4)$$

Recent studies into the fractal geometry characteristics of stream patterns have focused on the self-similarity of drainage networks with an eye

towards inferring optimal drainage networks and self-organization (Bak et al. 1987; Rodriguez-Iturbe et al. 1992; Maritan et al. 1996). Such interpretive work relies greatly on the quality and accuracy of the maps from which the fractal structure is inferred. The fractal dimensions of the study area are not explored here, but future research regarding the role of map scale in fractal geometry in the semi-arid Southwest is surely warranted.

This paper presents the results of a GIS-based watershed and drainage analysis in a semi-arid region in the U.S. Southwest. Stream channels were digitized and captured in GIS from a variety of data sources, including those represented on USGS 7.5' and 15' quadrangles and interpreted from large-scale aerial photography. The channels were ordered according to the Horton-Strahler technique; drainage characteristics extracted for 39 subwatersheds ranged in scale from less than 0.5 km^2 to almost 150 km^2. The scale and resolution of the data used to delineate stream channels significantly affected the ordering analysis as well as other geomorphic indices commonly used for watershed characterization and comparison.

Description of the Study Area

The USDA-ARS Walnut Gulch Experimental Watershed is located in southeastern Arizona and encompasses the historical town of Tombstone (Figure 1). The 148 km^2 watershed lies within the transition zone between the Chihuahuan and Sonoran deserts, and the climate is classified as semi-arid or steppe (Renard et al. 1993). Soils within the watershed are primarily gravelly sandy loams with a moderate to high calcium carbonate content. The majority of the watershed overlies deep alluvial deposits, but in some localized areas bedrock exists near the surface and exerts a controlling influence over the stream channel system. The drainage pattern is primarily dendritic with a relatively high density, but where near-surface bedrock or faults are exposed, the pattern is less dense and is more accurately described as rectangular. Eighty-nine recording rain gauges measure rainfall across 40 nested subwatersheds, many of whose runoff is gauged with either a supercritical flume, a v-notch weir, or a stilling well. Mean annual temperature in the town of Tombstone is 17.6°C and mean annual rainfall is approximately 320 mm.

Approach

Four sets of base map data, two sets of topographic maps and two sets of low-level aerial photography, were compiled for this study (Table 1). The smallest-scale topographic maps were 1:62,500 USGS maps derived from national high-altitude photography (NHAP) with a 1–2 m pixel resolution and compiled as 15' quadrangles (USGS 1999). USGS 7.5' topographic maps (1:24,000 USGS), widely available and common in hydrologic research, were used as well. These maps were also derived from NHAP photography, but were produced on a 1:24,000 scale. Low-level aerial photographs of Walnut Gulch at a scale of 1:24,000 (1:24,000 a.p.) were taken in 1989. These photographs have a 0.5-m pixel resolution and were orthogonalized and geo-rectified to remove distortion and ensure positional accuracy. Large-scale versions of these photos, blown up to 1:5,000, were used to delineate stream channels. The largest-scale data set was a compendium of 1:12,000 aerial photographs with approximately 0.3-m pixel resolution that were neither geo-rectified nor orthogonalized. Because of the distortion inherent in these photos, direct comparison through overlays or positioning of the channels was impossible.

For direct comparison it is vital that various data sources used in an interpretation exercise be orthogonalized and geo-rectified. Orthogonalization entails removing horizontal and vertical image distortion resulting from lens curvature and instrument quality. Geo-rectification is used to properly locate landscape features relative to one another within an image. However, general trends and significant differences can be inferred from data that are not of commensurate quality. In this study, three sets of base data were geo-rectified and orthogonalized, whereas the 1:12,000 aerial photographs were not; some error is therefore accepted in the determination of channel lengths and drainage density. Any errors due to distortion or misplacement will be most significant on smaller areas; as the study area is increased, small-scale variability has less impact.

Table 1. Data sources used in the delineation of stream channels.

Data Source	Scale	Pixel Size	Orth.	Geo.
1:12,000	1:12,000	0.3m	n	n
1:24,000 a.p.	1:24,000, 1:5,000	0.5m	y	y
1:24,000 USGS	1:24,000	1–2m	y	y
1:62,500	1:62,500	1–2m	y	y

Orth. = orthogonalized; Geo. = geo-rectified.
1:12,000 = 1:12,000 aerial photographs, 1:24,000 a.p. = 1:24,000 aerial photographs, 1:24,000 USGS = 7.5' USGS topographic maps, 1:62,500 = 15' USGS topo maps.

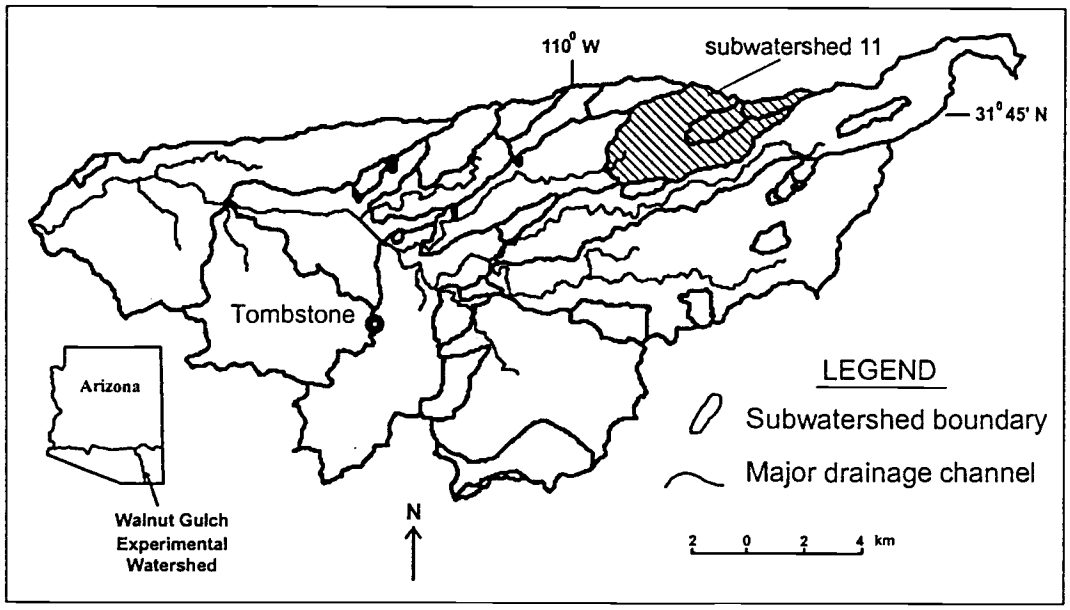

Figure 1. Location of the USDA-ARS Walnut Gulch Experimental Watershed showing nested subwatershed design with major stream channels. Subwatershed 11 is highlighted for reference in Figure 2.

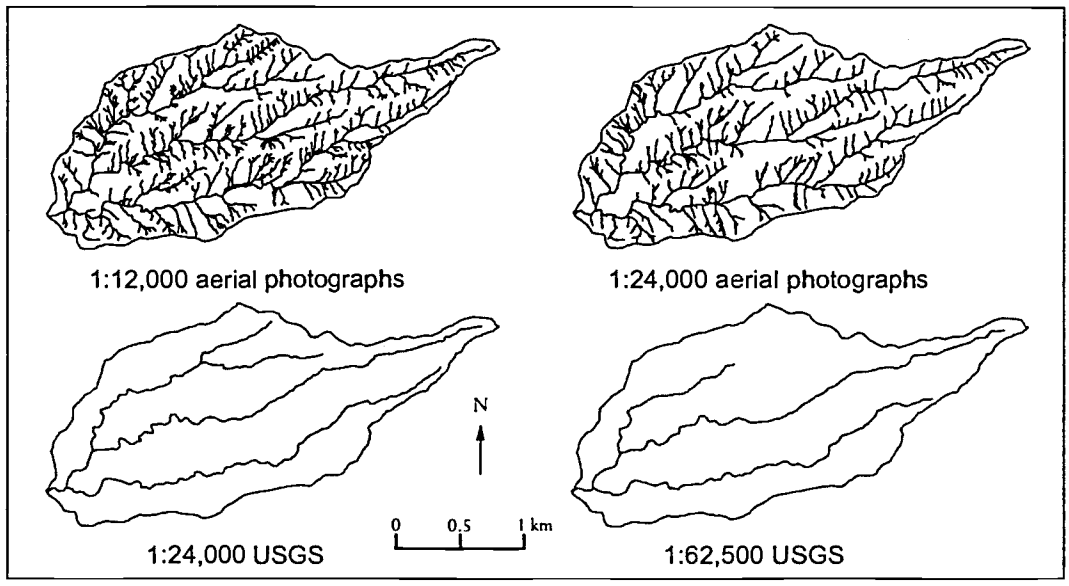

Figure 2. Digitized stream channels within subwatershed 11 on Walnut Gulch. Note how the drainage density decreases with decreasing scale while the underlying structure of the drainage pattern is retained.

Both of the USGS topographic map products are produced with defined stream channels; therefore, no interpretation as to channel position or length was made. Each of the blue lines designating a channel, including dashed lines indicating an ephemeral drainage, was digitized into Arc/Info GIS (names are necessary for factual reporting; however, the USDA neither guarantees nor warrants the standards of this product, and the use of the name by the USDA implies no approval of the product to the exclusion of others that may be suitable). Minor editing was done to ensure connectivity between quadrangles. Note that only larger channels are included in the USGS topographic maps due to the limitations imposed by national mapping standards; incipient drainages and small channels are generally not included on USGS maps. Determining channel locations from the aerial photography was a multistep process. In the case of the 1:12,000 photos, Wallace and Lane (1978) had previously traced the channel networks from a series of overlapping aerial photographs onto individual sheets. These tracings were digitized into Arc/Info, where some rubber sheeting was necessary to join and edge-match the sheets. Obvious position locators, such as channel intersections, were used to geo-position the digitized data. Because the 1:24,000 aerial photographs were geo-rectified and blown up to a scale of 1:5,000, no such data smoothing was necessary. Stream channels in each of the 24 photographs covering the watershed were traced onto mylar using a light table. Channels smaller than 1.5 m to 2 m in width were digitized as linear features, and larger channels were digitized as polygons, including alluvial features such as mid-channel bars and floodplains. Where channels were digitized as polygons, a perpendicular bisector was drawn using GIS techniques to extract a vector-based channel network. Figure 2 shows the digitized stream channels for subwatershed 11 for each of the base data sets.

A GIS algorithm for determining channel order presented by Miller et al. (1996) was used to order the channels for each of the four stream channel theme layers. This algorithm uses the arc-node vector topology defined for linear features in Arc/Info to sequentially sort through each of the channel segments and order them according to the Horton-Strahler technique.

The nested subwatershed design of the Walnut Gulch area served as the basis for investigation into issues of scale. The outlines of 39 subwatersheds ranging in size from 0.0035 km^2 to 148 km^2 were used to clip out the stream channels that were digitized from each of the data sources using GIS techniques. In this fashion, 156 stream channel GIS theme layers were created. Stream channel properties, such as maximum channel length, drainage density, and cumulative channel length, were determined using GIS analysis. Watershed area, used in the calculation of drainage density and in analyses of scale, was extracted from the GIS database for each of the 39 subwatersheds.

Results and Discussion

According to the largest-scale data used in this analysis, Walnut Gulch is a seventh-order watershed. However, as shown in Figure 3, ordering based on the other maps did not yield the same result; decreasing the map scale results in a decrease in the order of the watershed. An analysis of the four ordering systems using Horton's (1945) law of stream numbers (eq. 1) further illustrates the impact of map scale on ordering. In theory, a watershed should be represented by a single bifurcation ratio (R_B). Note that in Table 2 the bifurcation ratios are widely scattered, with a large decline in average R_B for the USGS data, indicat-

Table 2. Results of bifurcation ratio analysis.

	1:12,000		1:24,000 a.p.		1:24,000 USGS		1:62,500	
Order	No. of channels	R_B	No. of channels	R_B	No. of channels	R_B	No. of channels	R_B
1	6,286		3,572		205		74	
2	1,419	4.43	920	3.88	49	4.18	18	4.11
3	266	5.33	189	4.87	13	3.77	6	3.00
4	60	4.43	36	5.25	5	2.60	2	3.00
5	15	4.00	10	3.60	1	5.00	1	2.00
6	3	5.00	1	10.00				
7	1	3.00						
Mean R_B		4.37		5.52		3.89		3.03

ing that substantially fewer small channels were captured in the original interpretive work. The average R_B for the 1:24,000 a.p. data is somewhat deceiving because there are a large number of fifth-order channels and only one sixth-order channel. The high R_B between orders five and six is unexpected, but can be explained by the vagaries of aerial photo interpretation.

The answer to the question as to what constitutes a channel is both somewhat arbitrary and researcher dependent. A set of rules was drawn up by Miller (1995) for the determination of stream channels from the 1:24,000 aerial photographs, but these rules assuredly differ from Wallace and Lane (1978) and USGS methodologies. Random laboratory error cannot be ignored as a source of differences among maps, although quality control practices can reduce this problem. After the discrepancy in R_B values was found for the 1:24,000 a.p. and USGS data, a survey of the original photographs was undertaken to determine if an interpretive error caused the shift. It was found that a single channel, strategically placed, would cause a cascade in the ordering system and the R_B values similar to those of the 1:12,000 maps. However, according to the rules for channel interpretation, no changes could be made to the digitized data.

The average channel length of each order by map set was determined within the GIS. These data were used in the calculation of the length ratio (Horton's law of stream lengths, eq. 2). Results from this analysis were not as well behaved as those of the law of stream numbers (Figure 4). Although the 1:12,000 and 1:24,000 a.p. data progress in the expected geometric manner with increasing order, neither the 1:24,000 USGS nor the 1:62,500 maps exhibit as direct a trend. The differences in the USGS data relative to the aerial photography are due to the lack of small channels; the average first-order channels are more than a kilometer long on the 1:24,000 USGS data, compared to slightly over 200 m for the 1:24,000 a.p. data. The average length ratio decreases with scale: the 1:62,500 data have an average length ratio of 1.2, compared to a ratio of 2.2 for the 1:12,000 a.p. data. If the law of stream numbers were to be used as a proxy for watershed response, interpretation of runoff efficiency would be greatly affected by the underlying map scale.

As there is a direct relationship between channel order and watershed area, area may be substituted for channel order to represent increasing scale in channel investigations. This substitution reduces some of the influence of a researcher's tendencies in channel delineation because small discrepancies can lead to large differences in order analysis. Watershed area was used to illustrate the loss in small watershed variability with decreasing map scale and the strong relationship between channel length and watershed area.

Drainage density (R_D), a measure of the runoff and transport efficiency of a watershed, was also found to vary a great deal as a function of map scale. Decreasing the map scale resulted in a significant overall reduction in drainage density (Figure 5). Drainage densities were determined for each of the 39 subwatersheds used in this study, and their areas are plotted on the x-axis of Figure 5. Note the high variability in R_D at smaller watershed areas. This variability is due to localized differences in soils, topography, vegetation, and geology. Small-scale variability in channel definition is lost on the larger watersheds as average values tend to dampen smaller signals. Different erosional processes occur at different scales; at the finest scale, hillslope form is determined by sediment transport processes and by the production rate and availability of erodible materials. At progressively coarser scales, the watershed owes its form to a combination of processes operating on the finest scales and erosion and transport by smaller channels not depicted on the small-scale maps. Figure 5 underscores the need for detailed mapping and large-scale data on smaller watersheds. Note the dampening of the small watershed variability by the USGS data; many of the small, lower-order channels are not delineated on these maps, and the R_D values for those watersheds are therefore reduced. By removing the noise generated by small watershed variability, the overall picture of the watershed is significantly altered; here the USGS-based information appears to be more uniform across a range of scales than the aerial photography data.

The law of stream lengths can be used to illustrate the organization of stream patterns across a range of watershed scales: with increasing order (and therefore watershed area), there is a steady increase in both average and cumulative channel length (sum of all channel lengths within a basin). Linear regression models were fit between the cumulative channel lengths and watershed areas for the 39 subwatersheds at each of the map scales (Table 3). Strong power function relationships with high coefficients of determination (r^2)

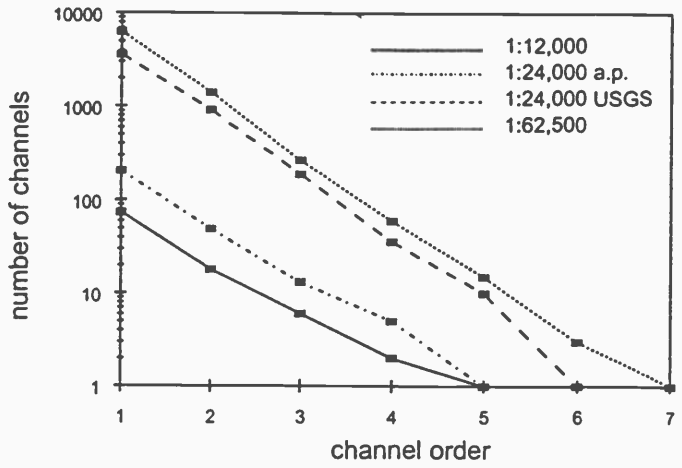

Figure 3. Number of stream channels by order as a function of map scale.

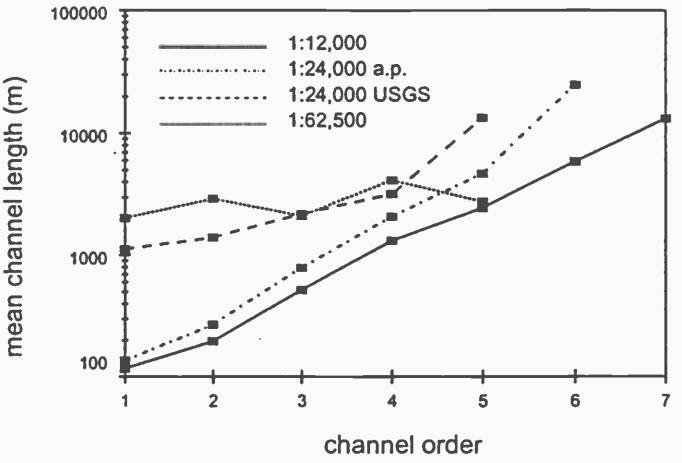

Figure 4. Average channel length by order as a function of map scale.

describe the increase in channel length with area (Figure 6).

All of the regression results presented in Table 3 were significantly different from one another. The principal separation between the results was the intercept, expressed as the coefficient in Table 3. The slopes of the lines, expressed as the exponent, are all similar. These results are perhaps indicative of the underlying drainage pattern structure and watershed routing efficiency.

Table 3. Linear regression models predicting cumulative channel length as function of watershed area expressed as power functions. Cumulative channel length = cA^b.

Map Source	Coefficient (c)	Exponent (b)	r^2
1:12,000	14.25	0.95	0.96
1:24,000 a.p.	10.93	0.92	0.89
1:24,000 USGS	3.99	0.93	0.97
1:62,500	2.11	0.95	0.86

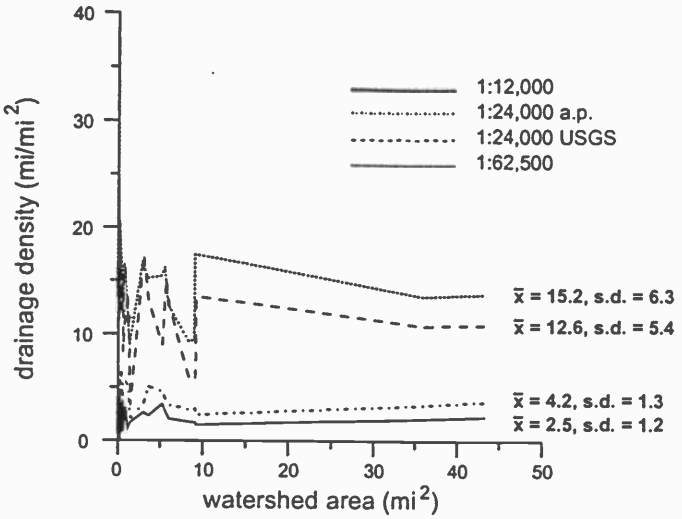

Figure 5. Drainage density as a function of watershed area and map scale.

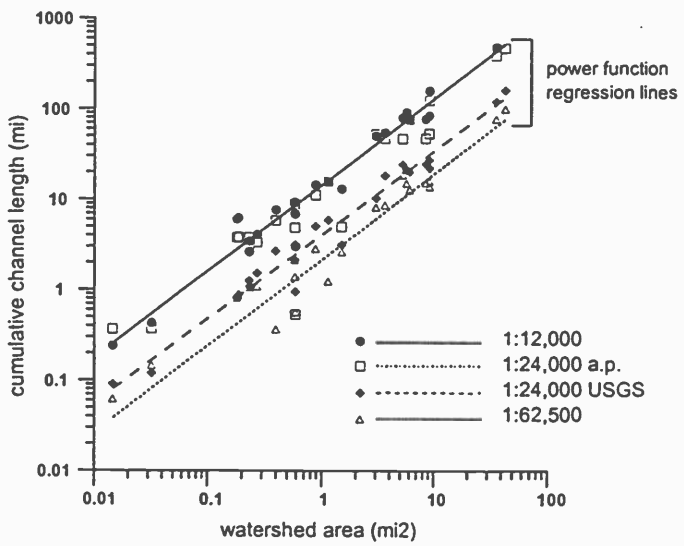

Figure 6. Cumulative channel length as a function of watershed area and map scale.

Conclusions

Four sets of maps and aerial photographs of different scales were used to delineate stream channels on the Walnut Gulch Experimental Watershed. Through GIS-based watershed and order analysis, it was demonstrated that the base scale significantly affects drainage articulation and basin characterization. Several reasons for differences in results are presented. The use of subjective guidelines for photo interpretation yields variable results even for the same base data. Base map scale restricts the amount of information that can be displayed on a given map. The original photo pixel resolution limits the amount of recorded information available for interpretation. When used for inter-basin comparison or watershed assessment, channel order analysis must be confined to maps constructed from similar data with similar techniques. Watershed area is perhaps better suited to studies involving scale issues in hydrologic and geomorphic analysis, and an approach for determining routing efficiency from watershed area is presented. Simple indices such as channel order are useful for watershed characterization and assessment, with the caveat that the techniques used to produce the results must be fully assessed before conclusions are drawn from the results.

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PRESCRIBED FIRE AND SIMULATED EROSION EFFECTS ON A SOUTHERN ARIZONA GRASSLAND: FIRST-YEAR RESULTS

Mary O'Dea[1] and D. Phillip Guertin[1]

Within an ecosystem, plant and soil processes interact to affect surface hydrology through their influence on surface roughness, soil structure, and evaporation, and through their relationship with soil biota. In the Southwest, decreases in perennial grass cover and erosion of uplands can initiate a decline in watershed condition. We know from both explicit research and intuition that the processes that characterize the watershed, such as geomorphology, hydrology, soil, and vegetation, are linked together. But what is not well known are the linkages and drivers that can advance the disruption of this integrated system.

The goal of this study is to identify and quantify the role of edaphic linkages between plant and surface hydrologic processes, specifically those that are dependent upon the physical, chemical, and biological features of the upper soil surface layer. The objective of this paper is to present first-year results in sediment yield, soil, and vegetation properties following a prescribed fire and simulated monsoon rainfall event on a perennial grassland community.

Materials and Methods

The study area is a small watershed located in Elgin, Arizona. This perennial grassland community contains native grasses, including plains lovegrass (*Eragrostis intermedia*), blue (*Bouteloua gracilis*), black (*B. eropida*), hairy (*B. hirusta*), and side-oats (*B. curtipendula*) gramas, other grama species, and *Muhlenbergia* spp. Mimosa (*Mimosa biuncifera*), rabbitbrush (*Chrysothamnus nauseosus*), various cacti, mesquite (*Prosopis velutina*), oaks, and juniper are also present.

The soil series at the site is a White House gravelly loam, whose parent material is an alluvium derived from andesite, rhyolite, limestone, and quartzsite. These soils are fairly deep with moderate to slow (0.06–0.2 in/hr) permeability and a high shrink/swell potential within the surface layer (SCS Soil Survey for Santa Cruz County, and parts of Cochise and Pima Counties, Arizona).

The study is a randomized complete block (block = replicate) with a split-plot design. Six blocks each containing four permanent runoff subplots were established, measuring 3 m wide by 10 m long with a 3-m buffer between plots. Each subplot was walled to disallow run on. Four treatments were randomly applied: an erosional rainfall event, a prescribed fire, a fire and erosion event, and a control. Twelve permanent sampling points were established systematically within each subplot.

A prescribed fire in early summer was applied following plot establishment and baseline data collection. The prescribed burn was planned for late spring–early summer to simulate the historic timing of fires in this area when lightning storms may occur.

Following the application of the burn treatment, assigned plots were forced into a visibly eroded state using a rotating-boom rainfall simulator. The assigned plots were treated at a rate of 2.5 $inhr^{-1}$ for 40 minutes; ponding occurred after 3.5 minutes. All sediment was captured in a pan at the foot of the plot.

One hundred percent of the vegetation within each quadrat was surveyed for frequency of species occurrence, basal diameter, and percent total canopy cover. In addition, total percent cover of cryptogams, rocks, annuals, and bare ground was also noted.

Samples were taken in the top 10 cm of soil because these layers are the most influential on infiltration rates in semi-arid soils (Dunne et al. 1991). Aggregate stability (Kemper and Rosenau 1986), particle-size distribution (Gee and Bauder 1986), and bulk density (Blake and Hartge 1986)

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were measured. A disk permeameter (infiltrometer) was used to measure changes in infiltration rates (Sullivan et al. 1996).

Tests for significant differences among treatments followed experimental design. Treatment effects on soil and plant parameters were analyzed with analysis of variance and covariance techniques.

Results and Discussion

Prior to treatment application there were no significant differences in mean percent cover for perennial grasses or herbaceous plants, cryptogams, rocks, or bare ground. Following the first growing season after treatment application, both perennial grass and herbaceous cover changed significantly ($p \leq 0.05$), reflecting treatment effects (Table 1). Significant decreases in perennial grass cover within the burned plots, and significant increases in herbaceous cover within the simulation and burn treatment, lead to questions regarding the stability of the highly disturbed communities (Figure 1).

There were also significant ($p \leq 0.05$) decreases in plant basal area among the burned plots (Table 1). There was little difference in basal area among the control and simulated erosion treatments. With the nearly 40 percent more total rainfall than the control plot, it was not unreasonable to expect a large accretion of biomass in the simulated erosion treatment. The large differences in basal area between the simulation and burn treatment with the other treatments illustrates not only the destructive nature of the treatment interaction, but also the variability among the treatments (Figure 2).

From field observation of the burned plots, the most notable characteristic of the perennial grass community is that both live and relic clumps remained on site following the first monsoon season. So although the relics may present physical barriers to overland flow this season, their presence in future years may be in question. These results therefore lead to questions regarding the possibility of an increased presence of annual species over the perennial ones within the highly disturbed sites, as well as a decrease in adequate barriers to overland flow in the future.

Following the first monsoon season there were no significant ($p \leq 0.05$) treatment effects on soil properties in either particle size distribution or percent stable aggregates (aggregate stability). Both the bulk soil and the sediment had highly variable particle size distributions, which included a large and variable clay fraction (30–60%). The number of percent stable aggregates (PSAs) did not significantly differ among treatments. Given that the PSAs were 0.3 mm or smaller, the wet-sieving method used to evaluate aggregate stability may not be appropriate to detect differences.

The burn treatments showed significant differences ($p \leq 0.05$) in both bulk density and surface infiltration rates compared to the unburned. Burned plots had significantly ($p \leq 0.05$) higher bulk density for the top 10 cm of the soil profile than the control. The simulation and burn treatment had significantly ($p \leq 0.05$) greater bulk density at the 6–10 cm depth compared to the surface 5 cm (Figure 3). Within the simulated erosion treatment, bulk density did not change significantly with depth. However, although the top 5 cm of the profile was not significantly different from the other treatments, the lower profile (6–10 cm) was significantly ($p \leq 0.05$) higher than the same depth in the control. Given the changes in the bulk density measurements it was not surprising to see similar patterns in the results for surface infiltration rates. As expected the burned treatments had significantly ($p \leq 0.05$) lower rates than the unburned treatments (Figure 4).

Changes in surface hydrology were expected, given the changes within the plant community and soil structure. Using total runoff and total

Table 1. First-year comparison of land cover among treatments.

	Percent Cover (%)					
Treatment	Basal Area (cm^2)	Perennial Grass	Cryptogams	Rocks	Herbs	Bare Ground
Control	254.8^{a}	20.4^{b}	0.5^{a}	0.7^{a}	0.9^{b}	77.5^{b}
Burned	185.3^{ab}	9.0^{c}	0.6^{a}	0.9^{a}	1.9^{ab}	87.7^{a}
Simulated erosion	319.8^{a}	29.8^{a}	0.5^{a}	0.8^{a}	1.9^{ab}	67.0^{c}
Simulation and burn	24.7^{b}	11.2^{c}	0.5^{a}	1.0^{a}	3.2^{a}	84.1^{ab}

Signifcant differences ($p \leq 0.05$) among means are denoted by different letters.

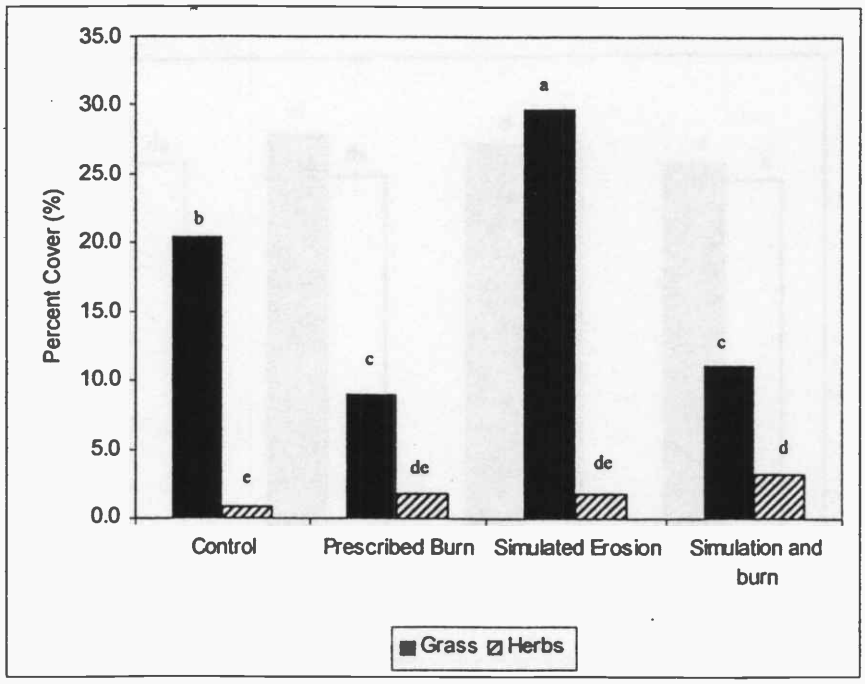

Figure 1. Comparison among treatments: Percent cover of perennial grass and annual herbaceous plants. 1 = Statistical comparison is within plant classes. 2 = Significant differences ($p \leq 0.05$) among means are denoted by changes in lettering.

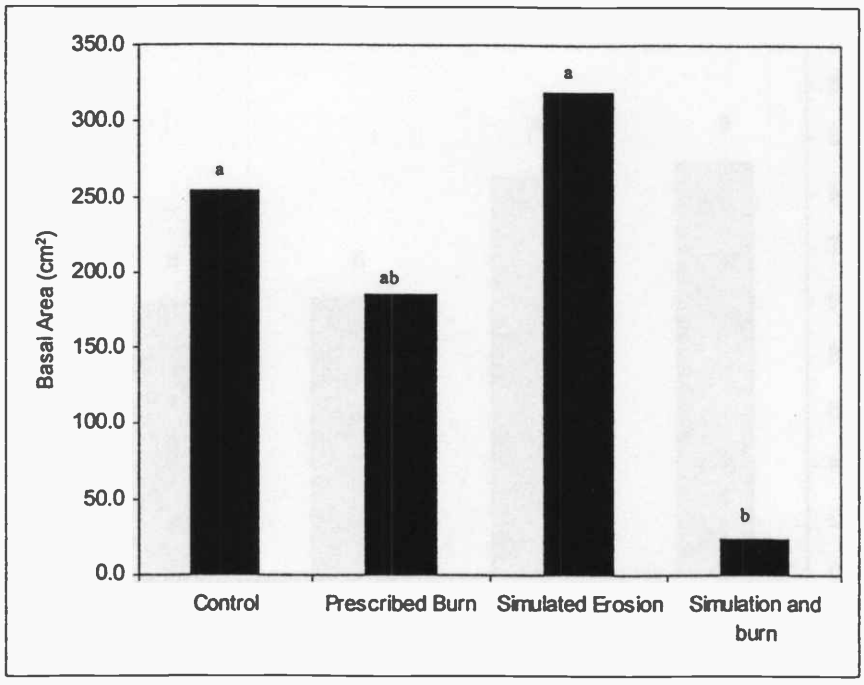

Figure 2. Comparison among treatments: Basal area of perennial grasses.

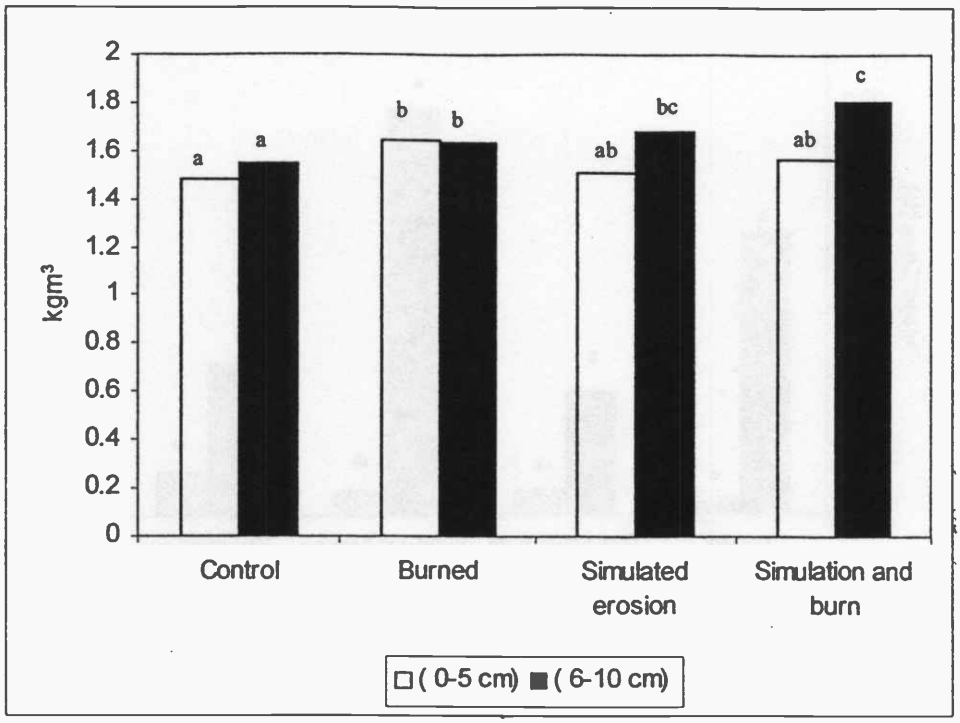

Figure 3. Comparison among treatments: Changes in bulk density with depth. Statistical comparisons are within depth classes.

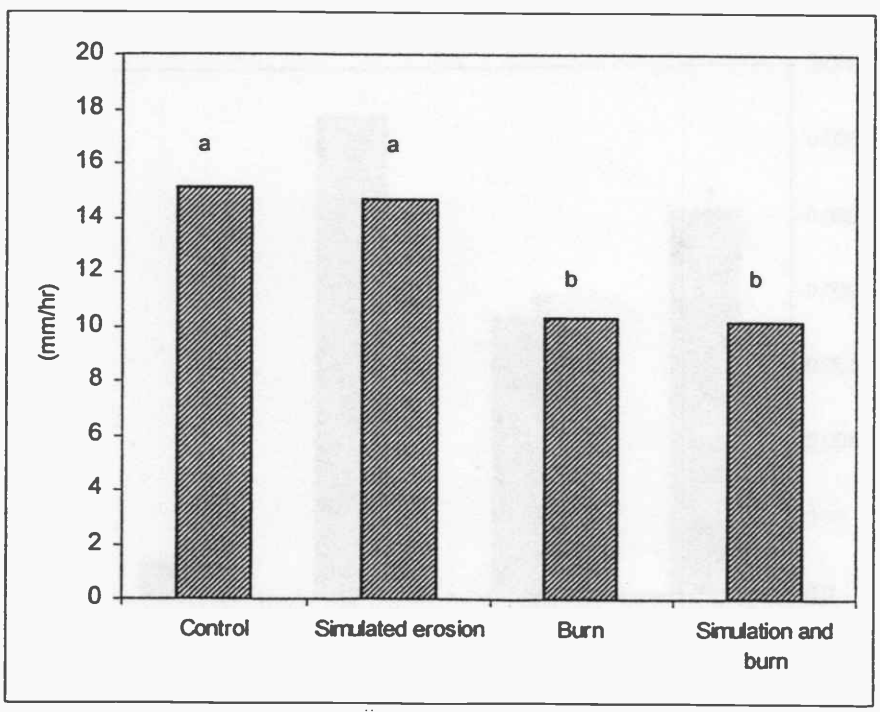

Figure 4. Comparison among treatments: Surface infiltration rates.

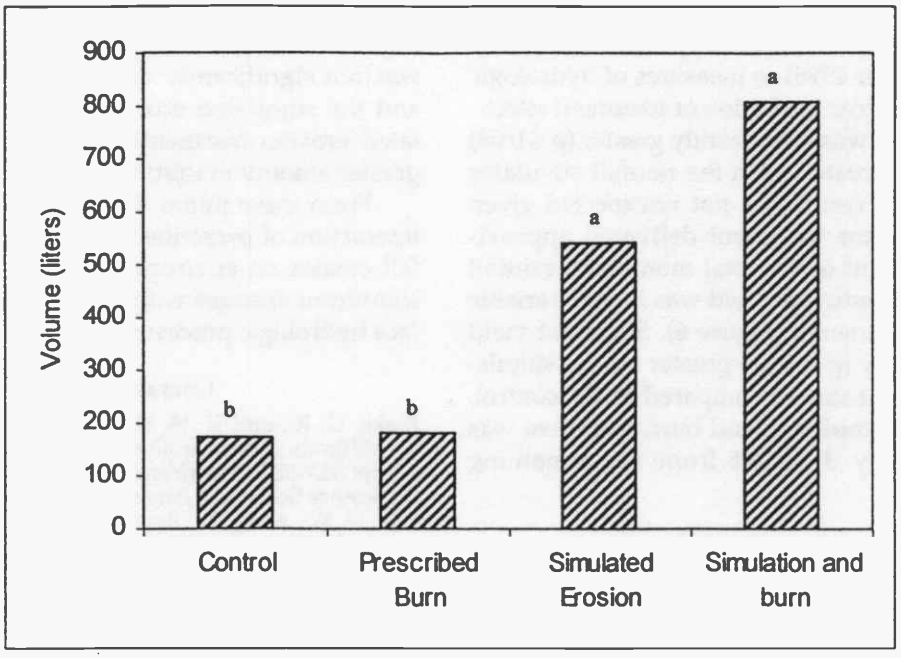

Figure 5. Comparison among treatments: Total runoff volume from 1 July 1998 to 1 October 1998 captured within the sediment pan. The values do not take into account losses due to evaporation or pan leakage.

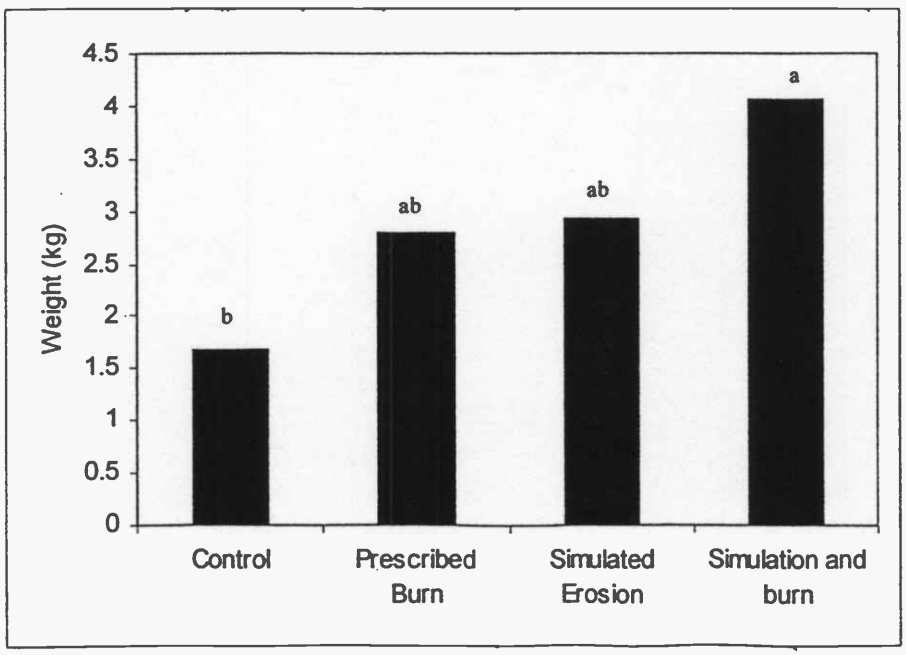

Figure 6. Comparison among treatments: Total sediment yield between 1 July 1998 and 1 October 1998.

sediment yield (in the sediment pans) captured during the monsoon season (approximately 1 July 1998 to 1 October 1998) as measures of hydrologic change gives a clear indication of treatment effect.

Total runoff was significantly greater ($p \leq 0.05$) from the plots treated with the rainfall simulator (Figure 5). This result was not unexpected given that the simulator treatment delivered approximately 40 percent of the total monsoonal rainfall for 1998. Total sediment yield was highly variable among the treatments (Figure 6). Sediment yield was significantly ($p \leq 0.05$) greater for the simulation and burn treatment compared to the control. However, the simulation and burn treatment was not significantly different from the remaining treatments.

Conclusion From First-Year Results

Burned plots showed the most dramatic changes in the first year following treatment application. These plots had lowered perennial plant cover and basal area, with a shift to increased annual cover. In addition, these plots had the highest values for bulk density, with the simulation and burn treatment having a significant change with depth. Moreover, these treatments displayed the slowest values for surface infiltration rates. Sediment yield was also greatest within the burn treatments. It is important to note that although the sediment yield was not significantly different between the burn and the simulated erosion treatments, the simulated erosion treatment had been subjected to a greater amount of rainfall (approximately 40%).

From these initial findings it appears that the interaction of prescribed fire and monsoonal rainfall creates an environment in which potentially significant changes within the plant, soil, and surface hydrologic processes may occur.

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RUCKER LAKE: A HISTORY OF RECENT CONDITIONS AFFECTING A SOUTHEASTERN ARIZONA WATERSHED

Robert E. Lefevre[1] and Daniel G. Neary[2]

As recently as 1970, there were 12 recreational lakes in southeastern Arizona. Today, due to the high cost of maintaining them, only six remain. One of these is Rucker Lake, which has been a recreation destination in the Chiricahua Mountains of southeastern Arizona since the 1930s (Figure 1). It is located 48 km (30 mi) north of Douglas, Arizona, and 80 km (50 mi) northeast of Bisbee. Even at the height of its popularity, it was only 1.3 ha (3.2 ac) in size, with a small campground on the west end. As is the case with all recreation lakes in southeastern Arizona, it is artificial, created by a small concrete dam. Because of the flashy hydrology and high sediment loads of drainages emanating from the Chiricahua Mountains, Rucker Lake acts as a sediment trap (Baker et al. 1995; Lusby and Hadley 1967; DeBano et al. 1996). Periodic maintenance in the form of dredging every 3 to 4 years has been necessary to keep the lake open.

Recently, Rucker Lake has not been maintained due to changes in the watershed condition and sediment load caused by the 1994 Rattlesnake Fire, and the rising cost of dredging. It is currently full of alluvial deposits and there is no open water. This paper documents the current status of Rucker Lake and will assist managers on the Coronado National Forest in making decisions about future management of the lake.

Rucker Canyon Watershed

The watershed is approximately 1902 ha (4700 ac) in size. Its vegetation consists of 51 percent transition coniferous forest, 3 percent mixed conifer and spruce-fir, 8 percent riparian, and 38 percent oak woodland. These vegetation types are described in the Coronado Land Management Plan (USDA Forest Service 1986), McLaughlin (1995), and Warshall (1995).

Broadleaf evergreen woodlands are found at elevations of about 1460 to 1645 m (4800–5400 ft). Mean annual air temperatures range from about 11.1° to 14.4°C (52–58°F). Mean annual precipitation ranges about 400–480 mm (16–19 inches). The dominant native vegetation is Emory oak (*Quercus emoryi*), Arizona white oak (*Quercus grisea*), alligator juniper (*Juniperus deppeana*), manzanita (*Arctostaphylos* spp.), and *Juniperus erythrocarpa*.

Transition coniferous forests are found at elevations of about 1980 to 2350 m (6500–7700 ft). Mean annual air temperatures range from about 9.4° to 12.7°C (49–55°F). Mean annual precipitation ranges from about 500 to 660 mm (20–26 inches). The dominant native vegetation is a mix of border pinyon (*Pinus discolor*), Chihuahua pine (*P. leiophylla*), ponderosa pine (*P. ponderosa*), manzanita (*Arctostaphylos* spp.), Arizona white oak (*Quercus grisea*), silver leaf oak (*Quercus hypoleucoides*), and alligator juniper (*Juniperus deppeana*).

Mixed conifer forests are found at higher elevations of about 2075 to 2750 m (6800–9000 ft). At those elevations, mean annual air temperature ranges from about 7.2° to 11.1°C (45–52°F). Mean annual precipitation ranges from about 560 to 660 mm (22–26 inches). The dominant native vegetation is ponderosa pine (*Pinus ponderosa*), alligator juniper (*Juniperus deppeana*), gambel oak (*Quercus gambelii*), and Douglas-fir (*Pseudotsuga menziesii*).

Spruce-fir coniferous forests are found at the highest elevations, about 2440–2990 m (8000–9800 ft). Mean annual air temperature at the mountain summits ranges from about 3.3° to 6.7°C (38–44°F). Mean annual precipitation ranges from about 762 to 890 mm (30–35 inches). The dominant native vegetation is white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), scattered aspen (*Populus tremuloides*), and in a few areas, high densities of Engelman spruce (*Picea englemannii*) and corkbark fir (*Abies lasiocarpa* var. *arizonica*).

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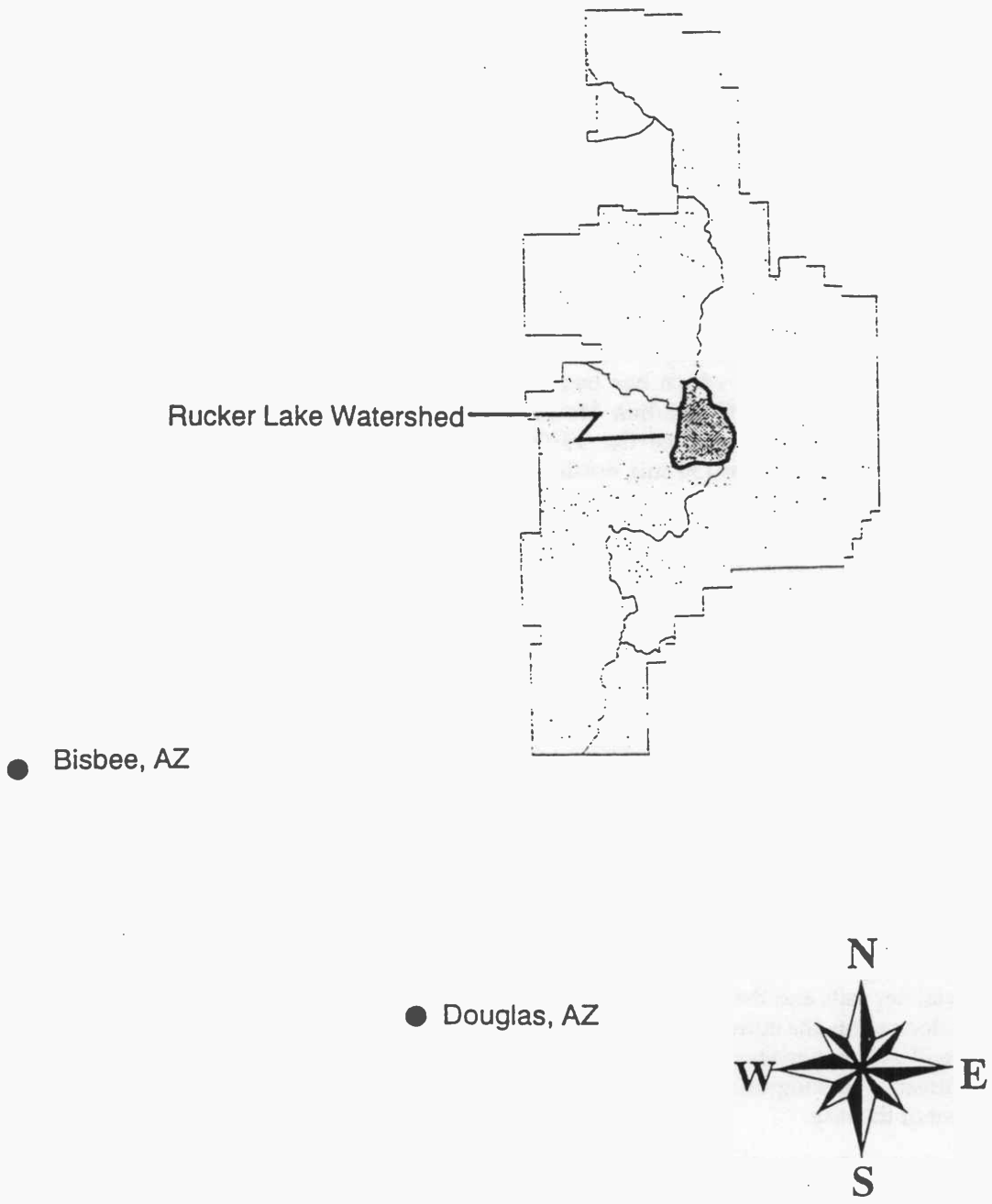

Figure 1. Rucker Lake watershed location, Coronado National Forest, Arizona.

The coniferous riparian forests are combinations of riparian obligate species in ponderosa pine–Douglas-fir forest and pine-oak-juniper woodland communities. They are found at elevations greater than 1585 m (5200 ft). The dominant native vegetation is Arizona sycamore (*Platanus wrightii*), Rocky Mountain maple (*Acer glabrum*), aspen (*Populus tremuloides*), ash (*Fraxinus* spp.), Arizona alder (*Alnus oblongifolia*), and Arizona cyprus (*Cupressus arizonica*).

History

Rucker Lake was originally known as "the Bathtub" when it was constructed in the 1930s by the CCC Administration (Newbury 1964). In 1947 it was raised to its present height by the sportsmen of Cochise County, and was commonly called Rucker Lake only after it became used for recreational fishing (Newbury 1964). Prominent citizens of Douglas and Bisbee worked to have an improved recreation area around the lake. At one time it was proposed that the lake be named Meisher Lake after W. H. Meischer, who is credited with being responsible for raising the dam to its present height (Newbury 1964).

Throughout the 1940s, 1950s, and early 1960s the Arizona Game and Fish Department stocked Rucker Lake around Easter and a fishing derby was held (personal communication, Medlock 1999 and Velasco 1999). It was a big event for the children of Douglas and Bisbee, who would line up along the bank waiting for the 9:00 a.m. starting whistle (personal communication, Medlock 1999). A prize was awarded for the biggest fish (personal comm., Velasco 1999). The fishing derby has ceased as an organized event, but stocking with catchable trout continued into the 1990s.

Ownership of the lake was turned over to the Forest Service from the Arizona Game and Fish Department by 1968. A water right for the lake was obtained by the USDA Forest Service in 1990.

Apparently sedimentation has been a problem since the early days. Records show that dredging of the lake had become a routine occurrence by 1963. In September, every other year, the gate valve in the dam was opened for about a week. Water and sediment were allowed to pass through the dam. Every third year, the Forest Service road crew brought in heavy equipment and dredged the lake, using the gravel to plate the parking areas and roads in Rucker Canyon (personal communication, Velasco 1999). Approximately 22,935 m^3 (30,000 yds^3) were removed with each full dredging.

In addition to the sedimentation problem, the lake had a leak. Dick Langford and other Forest Service veterans knew that water flowed through the gravel bench north and west of the lake. The presence of the dam merely provided a more continuous source of water for that flow (personal communication, Velasco 1999). The dam was treated with gunnite in 1964 in hopes of keeping water in the lake longer, but the leak through the gravel to the north remained. In 1966, employees of a mine in Bisbee were hired to pump a jelly substance into the bank along the north and west side of the lake in an attempt to seal the flow of water. It apparently was not successful. In 1970, the Forest Service dug a ditch 3 m (10 ft) deep along the north side of the dam and mixed bentonite into the soil. This was not completely successful either. Water can still be observed coming out of the ground in the middle of the road between Bathtub Campground and the confluence of Rucker Canyon and the North Fork of Rucker Canyon (personal communication, Velasco 1999).

Changes in dealing with the sedimentation problem began in 1977. The practice of opening the valve every other year ended because of concerns for the effects it had on habitat for the Mexican stoneroller (*Campostoma ornatum pricei*), a native fish that occupies the stream below the dam. In response to this change in management, six gabion basket retention dams were constructed upstream from the lake in 1979 to trap sediment. These were cleaned in 1980, but it was determined that the retention dams were too small to catch a significant amount of material and cleaning them did not remove all that was caught (Lefevre 1979; personal communication, Velasco 1999). Dredging of the lake was continued on a regular basis by contract and by joint projects with the Arizona Game and Fish Department in 1979, 1986, and about 1990.

The sedimentation problem continued, and became more of a burden each time the lake was dredged because of increasing costs. In 1982 and 1983 a watershed survey was conducted by the Forest Service to determine the source of sediment. The uplands were found to be heavily vegetated with nearly 100 percent ground cover on the forest floor with the exception of a number of rock slides. These rock slides were identified as the source of material to the channel. The channel itself was found to be in stable condition in spite of conducting large amounts of material. Flood-prone areas along the channel supported well-developed

stands of cypress, pine, and various species of oak. These flood-prone areas were assumed to be composed of gravel carried to the site during prehistoric events. In 1983, it was determined that not all those events were prehistoric. The trail in Bear Canyon (a tributary to Rucker Creek) shows evidence that large amounts of material have moved since construction of the trail. Blazes on trees that were put in place about 1.2 m (4 ft) off the ground are now ankle high.

1994—The Rattlesnake Fire

The Rattlesnake Fire burned about half of the 1900 ha (4700 ac) Rucker Canyon watershed during 1994. The fire area was not seeded artificially as part of a burned area emergency rehabilitation. However, the uplands were inspected for evidence of fire severity. Overstory vegetation throughout the transition coniferous forest had been killed by the wildfire. The pre-fire ground cover, which consisted mostly of forest floor litter, was reduced to zero. The pre-fire closed canopy had not allowed for much development of ground vegetation. Unlike most of the rest of the burned area of the Rattlesnake Fire, the Rucker Canyon watershed did not have a good seed source for reestablishment of ground vegetation.

Post-fire runoff was predicted to be dramatically higher than pre-fire conditions (Figure 2). The stream channel was inspected and appeared stable in the reach immediately upstream from the lake. There were no signs of large amounts of deposition except in the campground and lake. It was about time to dredge the lake again based on the history of dredging, but the sediment yield from the burned watershed was much greater than any previously witnessed. The lake filled in quickly and aggraded up the channel, completely covering the campground area with about 76 cm (30 inches) of new material.

1996—Recovery Processes Assessed

Alluvial material found in the channel appeared more angular than that found deposited in the lake. This would indicate that the material in the lake had been in the riparian and channel system a long period of time compared to that now found in the channel. Erosion in the burned area of the headwaters of the watershed obviously has provided a large source of fresh sediment. High streamflows have subsequently carried this material through the channel system and into Rucker Lake. It was estimated that more than enough material was stored in the channel to fill the lake with sediment again in a short period of time.

The "toe" of a fresh deposit of erosional material was found about 2.4 km (1.5 mi) upstream from the lake. An interpretation of this observation in 1996 was that material was moving through the system in waves. The channel between the point of fresh alluvium and the lake appeared to have few if any signs of newly deposited material, signifying that the worst sedimentation was yet to come.

The uplands were monitored with photographs, and recovery was found to be very slow. Gullying was observed in the upper watershed. Soil movement in the upper watershed was calculated using the Universal Soil Loss Equation. Losses in the burned area were estimated to be four times the amount in unburned areas. Delivery of sediment to the channel was estimated to be about three times that of unburned areas (Lefevre 1996, 1997).

In addition to the observations described above, a more detailed channel investigation was conducted in the vicinity of the Rucker Lake Campground, where new deposits had buried some picnic tables, tent sites, and campfire grills. It had been proposed to construct a dike to protect areas of the campground that had not yet been impacted from future sedimentation or channel encroachment. It was determined that the channel could be steered to the north side of the canyon for storms up to the 25-year event by constructing a dike 1 m (3 ft) tall. This evaluation was made assuming that the new deposits in the campground area were permanent, and that dredging them out would be futile (Lefevre 1996). Due to the cost of such a project, no dike was constructed.

1998—Upland Recovery Finally Begins

Observations in the upper watershed conducted in 1998 noted that ground vegetation had become established from seed blown in from adjacent slopes in the highest portions of the basin, and that trees had begun to fall, providing some protection to the soil surface. Also, the rate of gully formation appeared to be slowing. Midway down the slopes grass stands were becoming established, although fewer trees had fallen. The lowest portions of the burned area had not begun to revegetate significantly (Hocken 1998).

An observation of the channel conditions noted that the "gap" between the lake and newly deposited channel material had disappeared. Newly deposited material was found continuously from the lake upstream to the bend in the creek

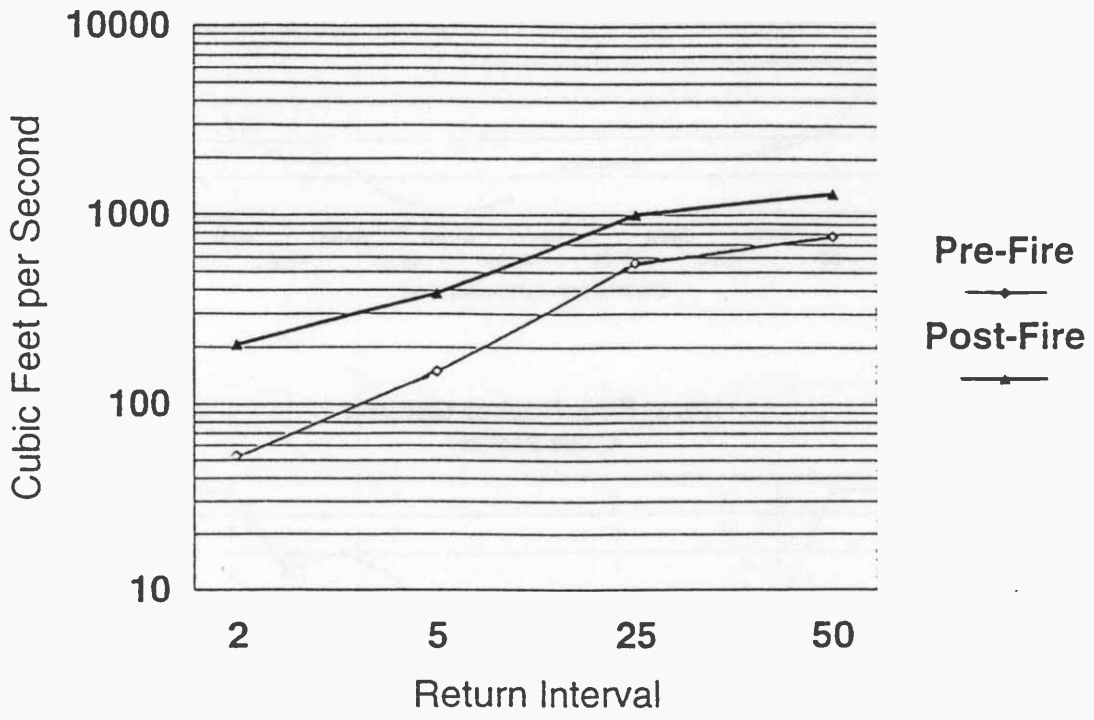

Figure 2. Predicted pre-fire and post-fire peak flows in Rucker Canyon.

and beyond. The significance of this observation was that no new wave of material appeared to be moving through the system.

The Current Situation

Channel cross sections and pebble counts in Rucker Canyon and the North Fork of Rucker Canyon (a nearly unburned tributary to Rucker Canyon) provide some important information about the current sedimentation situation (Figure 3). Cross-section data (Figure 4) indicate that in the burned watershed the entire flood-prone area is covered with water more often than the expected 1.5-year interval (bankfull discharge; Moody et al. 1998). The unburned watershed has the more desirable situation of bankfull flooding outside the channel occurring about every 1.5 years. For the Rucker Canyon Creek channel to stabilize, the banks will have to continue to build with available sediment. This should happen as fresh material is deposited on old terraces, resulting in a deeper channel. Observations of old terraces away from the active channel indicate that any new, post-fire terraces and freshly capped old terraces will become vegetated and stable.

Channel substrate is dominated by sand and gravel in Rucker Creek, and by cobbles in the North Fork of Rucker Canyon (Figure 4). Most exposed material in Rucker Canyon appears new by its color, but previous observations of more angular material in the channel than in the lake are no longer true. Apparently new material is being transported all the way to the lake and deposited on top of the older material. In addition, small-sized older sediments from the slopes and terraces of the channel are being mixed with new material throughout the channel. Because of the smaller size of the alluvial material currently in the channel of the burned watershed, much more sediment moves with every runoff event compared to the unburned watershed.

The situation of frequent flooding and small-sized material has resulted in terraces along the channel receiving additional sediment since the 1994 Rattlesnake Fire. These terraces have been in place for many years, as evidenced by the old trees

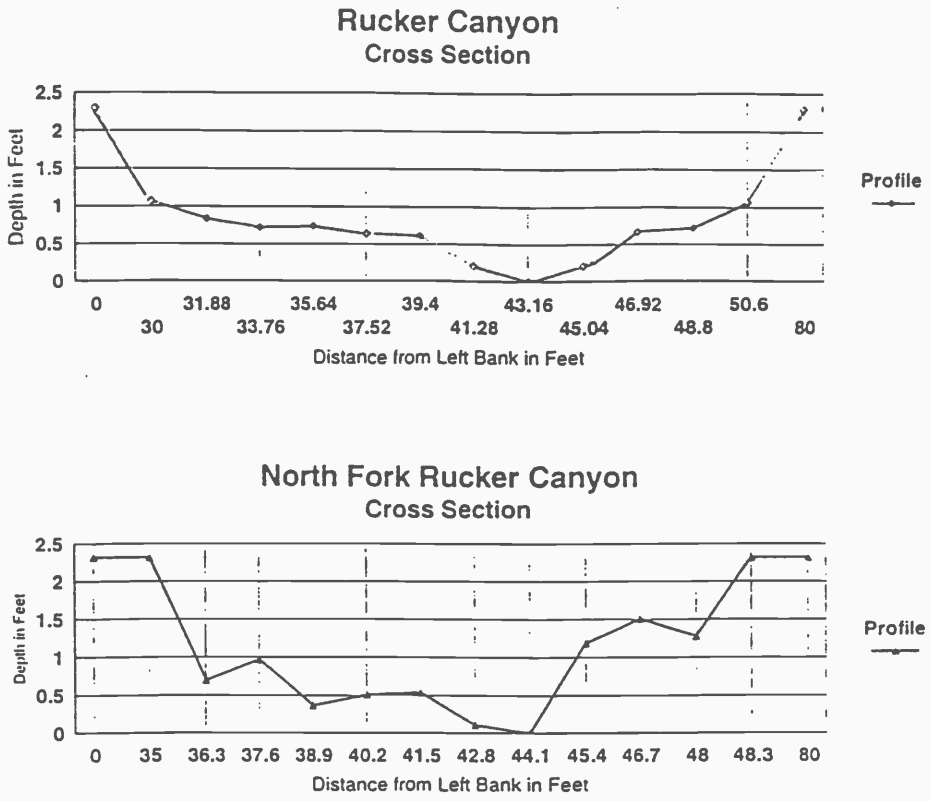

Figure 3. Cross-sections of Rucker Canyon Creek (burned) and the North Fork of Rucker Canyon Creek (unburned).

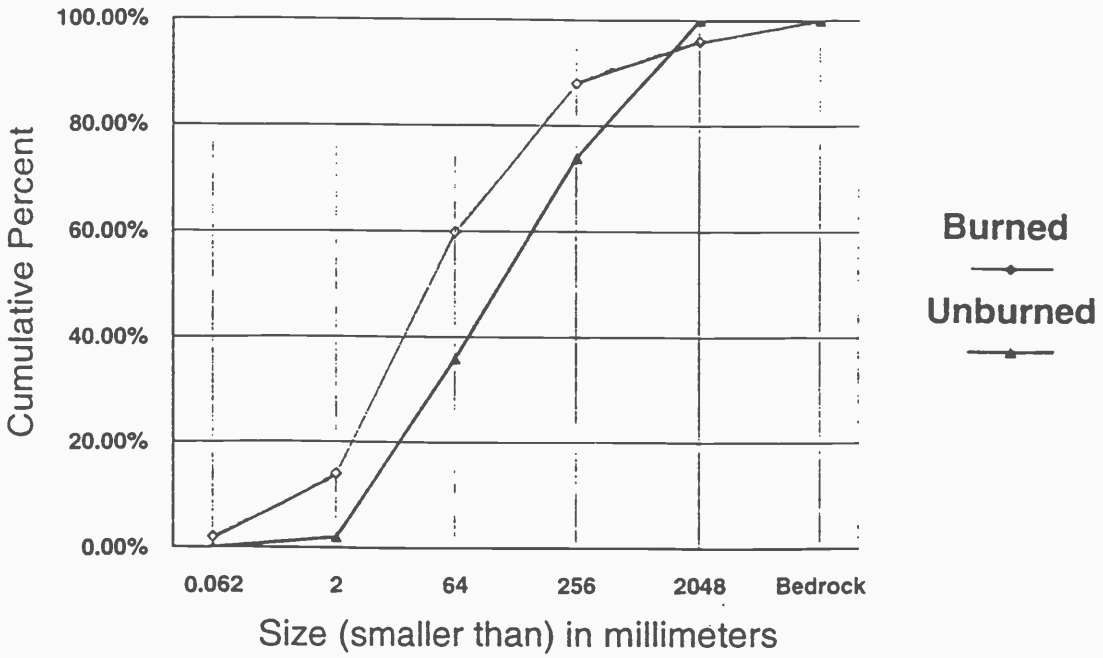

Figure 4. Stream substrate composition of Rucker Canyon Creek (burned) and the North Fork of Rucker Canyon Creek (unburned).

found on them. Future channel development will most likely consist of adjustments in depth and width to carry most flows within the confines of the new terrace situation. New terrace elevations should become stabilized with vegetation.

Observations were made in the winter of 1999 of two major channel reaches upstream from Rucker Lake. The channel conditions and trends are described below.

Rucker Creek Channel from the Campground Upstream to the Bend

The channel is formed on old alluvium similar to that brought in since 1994. Some new terraces have formed, and some old ones have received a veneer of new material. There is ample evidence of fresh erosional material being stored in the channel in lower-gradient reaches. This stored material is smaller and more mobile than the substrate of an adjacent unburned channel. Most of the stored material is behind temporary debris dams. These pockets of stored material may be stable for many years, allowing new vegetation to establish. However, when the material forming the debris dam decays or otherwise breaks down, the creek will downcut through the stored sediment, carrying much of it downstream. This process has apparently been happening for many years and is not something new caused by the Rattlesnake Fire.

Rucker Creek Channel Upstream from the Bend

The creek channel turns abruptly from the point where the trail leaves the canyon bottom. Upstream, the area is characterized as a bedrock-confined channel with vertical canyon walls. There is very little material stored in this reach, as its gradient is steep enough to produce the streamflow velocities needed to process all the sediment transported in from upstream. It stays scoured to bedrock. The channel is acting as a pipe, moving water and sediment debris efficiently from the uplands to the lower reach of the channel. As in the lower reach, there are a few debris dams storing material, but this process is insignificant compared to the role this reach plays in moving material. Observations were made at the mouths of all tributary channels and only two were found with any indications of residual sediment. Most were steep bedrock chutes. Apparently most of the fire-generated sediment material is transported very efficiently from the upland gully systems to the main channel of Rucker Canyon, and ultimately into Rucker Lake.

Rucker Lake Campground

The alluvium aggrading through the campground should be considered the new base level. Even if the lake is dredged, this new terrace will remain. Trying to dig out areas around the picnic tables and grills will be temporary because they will fill back in. If a campground is desirable here, the facilities must be on the new level. The same goes for a new road crossing to get from the main road to the campground site; the existing level of gravel should be considered a new base level.

Management Opportunities

The channel is quite efficient at transporting material from the upper watershed to the Rucker Lake area; when the upper watershed is stabilized at pre-fire conditions, it is likely that sediment loads will be similar to pre-fire conditions. Such conditions are expected 5 or 6 years after the fire, but may take much longer. The best circumstance expected will be that of a high sediment yield system similar to the situation before the Rattlesnake Fire. Rucker Lake could be dredged and maintained as a recreation facility at a cost similar to that experienced prior to the Rattlesnake Fire. That would mean dredging every 3–6 years.

The campground can be reconstructed at the new terrace level, but it must be recognized that the channel within the new terrace is not yet stabilized. It may continue to move laterally through the new deposits until it reaches an appropriate course for the gradient and sediment load.

Conclusions and Implications for Other Areas

There are other recreation lakes in southeastern Arizona and over the years there have been proposals for more recreation lakes. The observations made in the Rucker Lake case are that such lakes carry high maintenance costs due to the naturally high sediment loads characteristic of the steep mountain watersheds in southeast Arizona. Rucker Lake essentially functions like a debris basin. These structures are very efficient at trapping sediment but they have a high maintenance cost. Changes in upland watershed conditions induced by wildfires will disrupt maintenance schedules and temporarily increase costs.

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POST-FIRE BURNED AREA EMERGENCY REHABILITATION TREATMENTS: DO THEY WORK IN THE SOUTHWEST?

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A number of rehabilitation techniques have been used after wildfires to stabilize slopes, channels, and roads, to reduce erosion, or to improve watershed function. A West-wide project was started in 1998 to evaluate burned area emergency rehabilitation (BAER) projects on U.S. Forest Service lands. The Southwest is of special concern because the spring fire season is often followed by heavy monsoonal rainfall.

Recent large, high-severity fires in the Southwest and subsequent hydrological events have generated renewed interest in the linkage between fire and on-site and downstream effects. Fire is a natural and important part of the disturbance regime of forest ecosystems. The suppression of fires in the Southwest, beginning with grazing in the late 1800s, has altered natural fire occurrence. Later in the twentieth century, fire suppression allowed fuel loading and forest floor material to increase, resulting in more intense and larger fires (Neary et al. 1999). These high-severity fires are of particular interest because they can have a significant effect on soil productivity, watershed response, and downstream sedimentation. During severe fire seasons, the Forest Service and other land-management agencies spend millions of dollars on post-fire emergency watershed rehabilitation measures intended to minimize on-site erosion and off-site sedimentation and hydrologic damage.

Burned Area Emergency Rehabilitation (BAER) Program History

Emergency watershed rehabilitation after wildfires was first reported in the late 1960s and early 1970s. At that time, aerial seeding was the most common rehabilitation treatment. No formal rehabilitation program existed, so rehabilitation funds were obtained from fire-suppression accounts. In the early 70s, Congress conducted an inquiry to determine the need for and use of emergency watershed rehabilitation funds. Because of questions raised in that inquiry, the BAER program was started in 1974.

In the late 1980s, after the Yellowstone fires, a coordinated interagency effort was initiated to train BAER team leaders. A program was started to coordinate evaluation of fire severity, funding request procedures, and treatment options. Annual BAER training programs at regional and national levels were also initiated.

In 1987 several large fires in California caused expenditures for BAER treatments that exceeded previous annual BAER budgets. On several occasions requests were made for items that were considered "non-emergency," so BAER team leaders were trained to discern what constituted a real emergency and therefore what warranted rehabilitation treatments. During that time, National Forest specialists were encouraged to do implementation monitoring as well as some form of effectiveness monitoring. In the mid 1990s a major effort was undertaken to revise and update the BAER handbook. A steering group consisting of regional BAER coordinators and other specialists organized and developed the handbooks that are in use today. Because individual agencies have interpreted the congressional appropriations for BAER differently, an effort was made to standardize BAER policies across various federal agency boundaries.

Three BAER program areas were targeted for improvement in the late 90s: increased training, policy consistency, and funding review. The three areas of training that were subsequently enhanced were BAER Team Leader training, implementation training, and on-the-ground treatment installation training. Courses were developed for the first two

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training needs but not the last. By 1999, the Forest Service, the Bureau of Land Management, the National Park Service, the Fish and Wildlife Service, and the Bureau of Indian Affairs had approved a policy for a consistent approach to BAER.

The new BAER policy broadens the scope and application of BAER analysis and treatment. The changes include (a) monitoring to determine if additional treatment is needed, (b) evaluating treatments to improve effectiveness, (c) repairing facilities for safety reasons, (d) stabilizing biotic communities, and (e) preventing unacceptable degradation of critical cultural sites and natural resources. Funding requests require careful scrutiny to ensure that they are reasonable, practicable, and cost-effective, and that they provide significant improvement over natural recovery.

The current Forest Service Manual (FSM 2523) and the Burned Area Emergency Rehabilitation Handbook (FSH 2509.13) require an immediate assessment of site conditions following wildfire. Where necessary, emergency rehabilitation measures are implemented to (a) minimize the threat to life and property on or off the site, (b) reduce the loss of soil and on-site productivity, (c) reduce adverse changes in streamflow regimes, and (e) reduce deterioration of the water quality.

Problem Statement and Objectives

The effectiveness of many emergency rehabilitation methods has not been systematically tested or validated. Although BAER expenditures accounted for less than 1 percent of total Forest Service fire expenditures in 1994 (Figure 1), concerns have been raised at a national level due to the rapidly rising costs of this program in the 1990s. BAER team leaders and decision makers often do not have the necessary information available to thoroughly evaluate the short- and long-term benefits (and costs) of various treatment options. In 1998, a joint study was initiated between the Rocky Mountain Research Station and the Pacific Southwest Station to evaluate past usage of emergency rehabilitation methods.

The objective of this West-wide study was to address the following concerns about post-fire rehabilitation methods:

- Effectiveness of rehabilitation treatments at reducing erosion or other effects
- Effectiveness of rehabilitation treatments in mitigating downstream effects
- Rehabilitation treatments effects on natural ecosystem recovery
- Hillslope and channel treatment comparisons in terms of relative benefits, and improvements over no-treatment options
- Information on the economic, social, and environmental costs and benefits of various rehabilitation treatments
- Ease of transfer of knowledge on treatment effectiveness from one location to another
- Information gaps needing further research and evaluation.

The BAER treatment assessment was undertaken to collect information about past use of BAER treatments, what made the treatments succeed or fail, and whether the treatment achieved the desired goal. Because much of the information was not published and was qualitative in nature, a survey was designed to obtain answers from resource specialists about specific questions regarding their BAER programs. Some of the results pertaining to the Southwest are reported herein.

Fire Impact on Ecosystems

All disturbances produce impacts on forest ecosystems. The level of impact and its direction (negative or positive) depend on ecosystem resistance and resilience as well as the severity of the disturbance. *Severity* is an important fire concept to consider because it occurs in a spectrum that may or may not occur entirely on the same site (DeBano et al. 1998). The variability in resource damage and response from site to site, fire to fire, and ecosystem to ecosystem is highly dependent on fire severity. As a physical-chemical process, fire produces a range of severities depending on the interactions of unit area energy release during burning (intensity), duration, fuel loading and combustion, vegetation type, climate, slope, topography, soil, and area burned. Intensity has often been confused with severity in BAER documentation. Severity is basically a qualitative measure of the effects of fire on soil and site resources, although some aspects can be quantified (Hartford and Fransen 1992).

Soils are critical to the functioning of hydrological processes. On a watershed basis, possible sediment increases and water yield responses to prescribed fires and wildfires are often a function of fire severity and the occurrence of hydrologic events. When examining a wide range of fire severities, impacts on hydrology and sediment loss can be minimal in the absence of an immediate precipitation event. However, a precipitation event following a large fire of moderate to high

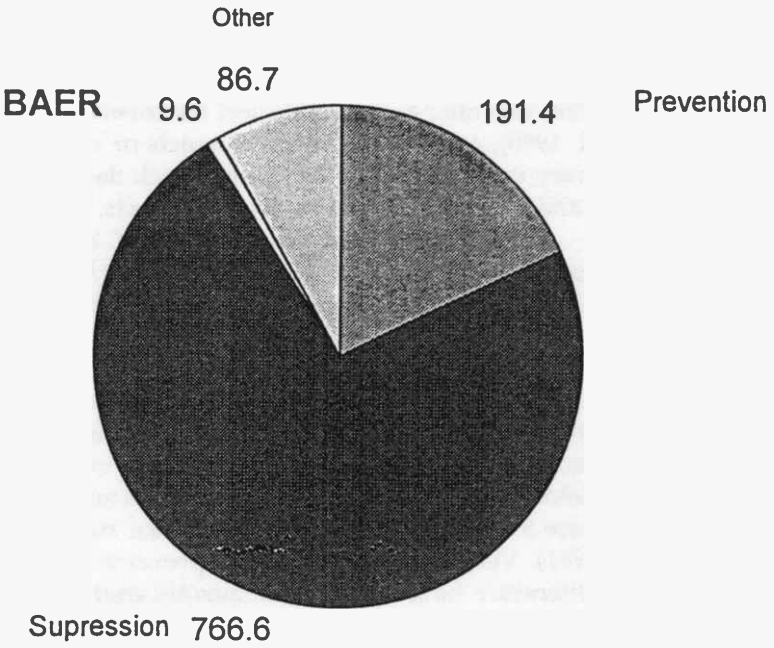

Figure 1. 1994 USDA Forest Service fire expenditures—1995 dollars in millions (from Schuster et al. 1997).

severity can produce far-reaching impacts. For example, increased runoff, peakflows, and sediment delivery to streams can impact fish populations and habitat (Rinne 1997). BAER treatments are designed to mitigate the impacts of severe wildfires on watershed condition.

Watershed condition describes the ability of a watershed system and soils to receive and process precipitation without ecosystem degradation. In good watershed conditions, precipitation infiltrates into the soil, and baseflows are sustained between storms. Except in the largest of precipitation events, rainfall does not significantly contribute to erosion because it does not flow over the surface where it can detach and transport sediments. Watersheds in good condition with minimal areas of exposed bedrock, roads, or compacted soils do not respond rapidly to most precipitation events. With poor watershed condition, rainfall runs quickly over the surface of the soil and there is little or no perennial baseflow between storms. Erosion is considerable during stormflows. Because of climate, soils, and vegetation, some watersheds, such as those in parts of the arid Southwest, may always function in this manner. Such watersheds are highly responsive in nature, being prone to flash floods.

Fire can destroy the organic forest floor and vegetation, and can alter the infiltration and percolation capacity of the soil by exposing it to raindrop impact or by creating water repellency (DeBano et al. 1998). This alters watershed condition, with erosion increasing as watershed condition deteriorates from good to poor. Loss of soil from hillslopes produces several significant ecosystem impacts. Soil movement into streams, lakes, and riparian zones not only deteriorates water quality, but can also change the geomorphic and hydrologic characteristics of these systems. More important, soil loss from hillslopes results in reduced future ecosystem productivity. BAER treatments are designed to mitigate these effects by improving post-fire watershed condition.

Erosion occurs when sediments are exposed to water or air and velocities are sufficient to detach and transport the sediments. It is a natural process occurring on landscapes at different rates and scales depending on geology, topography, vegetation, and climate. Fires and fire-suppression activities can affect erosion processes. Erosion is certainly the most visible and dramatic impact of wildfire apart from the consumption of vegetation (DeBano et al. 1998).

Fire-related sediment yields vary from ecoregion to ecoregion depending on factors such as fire frequency, climate, and vegetation, and geo-

morphic factors such as topography, geology, and soils (Swanson 1981). In some ecoregions more than 60 percent of the total landscape sediment production over the long term is fire related. Much of that sediment loss can occur the first year after a wildfire (Agee 1993; DeBano et al. 1998); consequently, BAER treatments can be very important in minimizing damage to both soil and watershed resources.

Post-wildfire sediment yields can range from very low in flat terrain and in the absence of major rainfall events to extreme in steep terrain affected by high-intensity monsoonal thunderstorms. Erosion in burned areas usually declines in subsequent years as the site stabilizes, but the rate varies depending on fire severity (DeBano et al. 1998). Nearly all fires increase sediment yield but it is the wildfires in steep terrain that produce the biggest amounts of sediment (Swanson 1981). Very few studies of wildfire effects in the literature have examined the effects of BAER treatments on sediment yield. One study in the Southwest Plateau and Plains Steppe and Shrub Province of Texas reported a five-fold reduction in sediment loss with the use of grass seeding (Wright et al. 1982).

BAER Treatments

BAER treatments have traditionally been grouped into hillslope, channel, and road treatments. Functionally, hillslope treatments can be divided into revegetation, erosion barrier, physical soil manipulation, and other ground cover treatments. Channel BAER treatments consist of check dams, grade stabilizers, and other miscellaneous treatments. The functional groups for road treatments are culvert-bridge-ditch improvements, shaping, protection, and surface manipulations. Another group of BAER treatments can be classified as treatment protection, including fencing, road or trail closures, and signing.

Hillslope treatments are designed to either reduce erosion or hold soil and sediments on the site (Miles et al. 1989). Revegetation treatments to aid plant reestablishment on burned slopes consist of aerial and ground seeding, fertilization, and mulching (with or without seeding). Erosion barriers such as logs (known as LEBs or log erosion barriers), straw wattles, straw bales, soil and sand bags, and silt fences are placed to trap eroded soil material on the slope. Physical soil manipulations such as contour trenching and ripping or tilling are used to trap and store eroding soil material or reduce erosion potential by improving surface roughness, depression storage, and infiltration capacity. Other ground cover techniques used to reduce erosion include slash spreading, felling snags along the contour, and laying erosion control fabric.

Channel treatments are designed to store sediment in channels or reduce the erosive power of water flow. Check dams are often used to detain sediment in channels. These can be made of straw wattles, straw bales, logs, rocks, or rock gabions. Grade stabilizers that lower the velocity of streamflow by reducing channel gradient via a series of steps can be constructed of logs or rocks. Other types of channel BAER treatments include debris basin construction, woody debris removal, small dams, and channel-bank armoring.

Road treatments are mainly used to help roads and road structures survive the additional streamflows and surface runoff that often occur after wildfires. Improvements such as culvert upgrades, culvert removals, riser installation, ditchline debris removal, and ditch check dam construction help control additional water flow or prevent ditch downcutting and culvert blowouts. Shaping of road surfaces by outsloping, water bar installation, and rolling dip construction all limit erosion by reducing water flow. Protection of roads can be accomplished by installing racks to trap trash and woody debris that might block culverts, patrolling roads and culverts during storm events, and constructing overflows that provide relief to culverts during excessively high stormflows. Surface manipulations such as resurfacing, ripping or tilling, and the armoring of crossings provide additional reductions of road surface and side-slope erosion during storm runoff events.

Other treatments used to aid post-wildfire watershed restoration include temporary fencing, road and trail closures, and signing. They can be used on hillslopes, in channels, or on roads. The purpose of these treatments is to keep vehicle, foot, and domestic animal traffic off sensitive, fire-disturbed soils and road and trail surfaces. Site vegetation and soil recovery often occurs faster if post-fire disturbances are reduced or eliminated.

BAER Treatment Effectiveness in the Southwest

Although there has been a lot of monitoring of BAER treatment implementation in the Southwest, effectiveness monitoring and evaluation data are very scarce. Some assessments of BAER treatment effectiveness have been reported by Anderson and Brooks (1975), Barro and Conard (1987), Beyers et al. (1998), Collins and Johnston (1995), Gautier

(1983), Miles et al. (1989), Taskey et al. (1989), and Wohlgenmuth et al. (1998). The three most common BAER treatments used on Forest Service lands in the western United States in the past 3 decades have been aerial seeding, contour felling, and mulching.

Historically, the Southwest Region (Region 3) has expended the least amount of BAER funds of the six western regions on aerial seeding and contour felling (Figure 2). Region 3 has ranked intermediate in the use of mulching. The scarce information available on aerial seeding, primarily from limited research studies, indicates that seeding in chaparral does little to reduce sediment loss the first year after wildfire (Beyers et al. 1998). In the two available studies, results were mixed and most of the success in reducing sediment loss from chaparral ecosystems occurred in the second and third years. In grass ecosystems of the Southwest Plateau and Plains Steppe and Shrub Province in Texas, Wright et al. (1982) reported a fivefold reduction in sediment loss with the use of grass seeding. Aerial seeding in the Southwest appears to be more effective in reducing soil loss in more mesic ecosystems, but much more information is needed to support this hypothesis. Additional monitoring or research studies are needed to get a better assessment of the effect of BAER treatments on both short- and long-term watershed restoration.

Because of the lack of substantial effectiveness monitoring, BAER team leaders and soils and watershed staff in the Southwest Region of the U.S. Forest Service were interviewed to determine, in the collective experience of these personnel, the most effective hillslope, channel, and road treatments. Each interviewee was asked to select the top three BAER treatments in each category.

Due to variable conditions of wildfire severity, vegetation, soils, geology, climate, individual experience, and so forth, no single treatment received a majority vote (Table 1). This fact most likely reflects the individual experiences of the BAER experts across the diverse landscape and ecosystems of the Southwest rather than any documented differences in treatment methods. The top-ranked BAER treatments were straw-bale check dams, LEBs–contour felling, seeding-fertilizing, culvert upgrading, and installing rolling dips and water bars.

Straw-bale check dams help detain sediment and reduce channel scour immediately upon

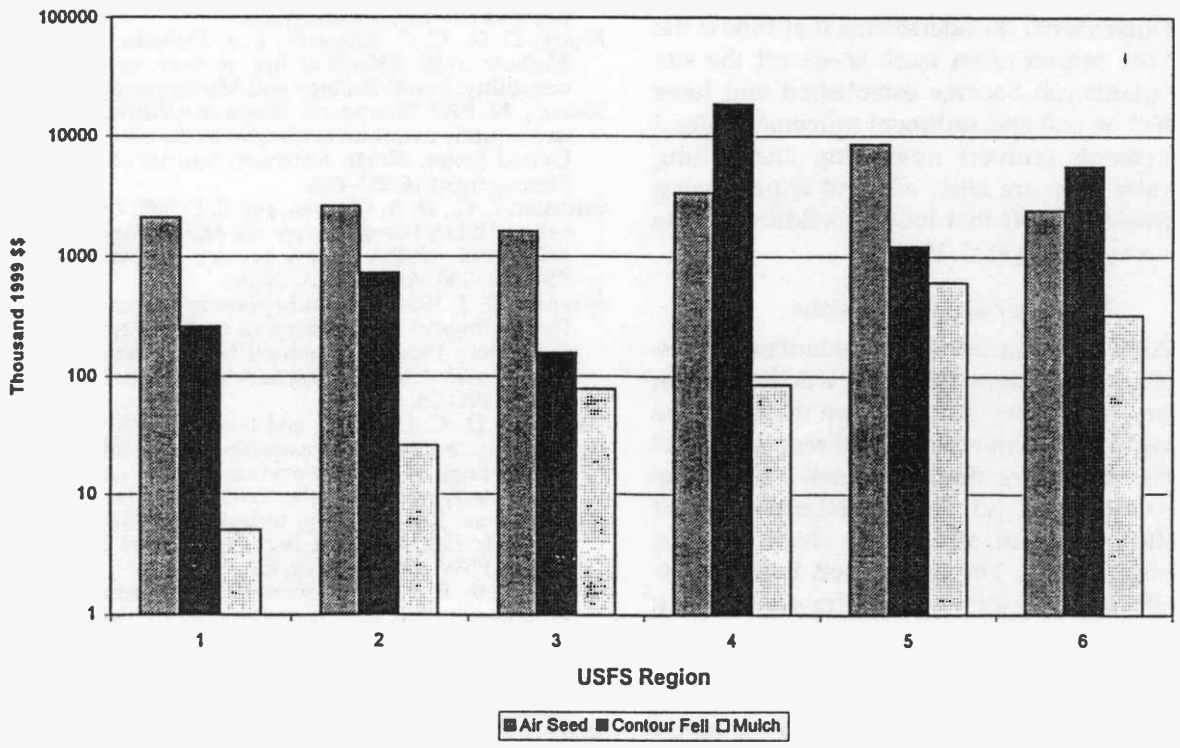

Figure 2. Cumulative BAER expenditures by USDA Forest Service region.

Table 1. National Forest best BAER treatment effectiveness, Southwest expert interviews (n = 12)

BAER Treatment Type	Treatment Effectiveness Response (%)
Hillslope	
1. LEBs–contour felling	30
2. Seeding-fertilizing	28
3. Slash scattering	20
4. Temporary fencing	16
Channel	
1. Straw-bale check dams	35
2. Channel debris clearing	10
3. Log grade stabilizers	10
4. Rock gabions	10
Road	
1. Culvert upgrading	22
2. Rolling dips–water bars	21
3. Trash racks	13
4. Outsloping roads	7

installation. Log erosion barriers–contour felling can also show immediate results if installed properly. Seeding-fertilizing has a mixed record. Post-fire seeding frequently gives poor results because seed rarely has time to germinate. The intense monsoonal thunderstorms that follow the spring fire season often wash seeds off the site before plants can become established and have any effect on soil and sediment movement. Road improvements (culvert upgrading and rolling dips–water bars) are fairly efficient at processing the increased runoff that follows wildfires in the Southwest (DeBano et al. 1998).

Summary and Conclusions

The BAER program involves immediate assessment of site conditions following wildfire. When necessary, emergency rehabilitation measures are implemented to (a) minimize the threat to life and property, (b) reduce the loss of soil and on-site productivity, and (c) reduce deterioration of watershed condition, streamflow characteristics, and water quality. The top-ranked BAER treatments in the Southwest are (a) straw-bale check dams, (b) LEBs–contour felling, (c) seeding-fertilizing, (d) culvert upgrading, and (e) installing rolling dips and water bars. Additional monitoring or research studies are needed to get a better understanding of the effectiveness of BAER treatments on watershed restoration.

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FIRE-INDUCED WATER REPELLENCY IN SOILS: HYDROLOGIC IMPLICATIONS

Leonard F. DeBano

Water-repellent soils produced during fires in wildland ecosystems have attracted the attention of soil scientists and hydrologists throughout the world for more than 3 decades. The results of numerous studies have related fire-induced water repellency to specific hydrologic processes such as infiltration, runoff, and surface erosion (rill and sheet erosion). Relating fire-induced water repellency to overall watershed performance in terms of streamflow responses, however, has been more difficult. This paper reviews the formation of fire-induced water-repellent soils, the effect of soil water repellency on infiltration and runoff, erosional processes unique to water-repellent soils, and watershed and plot studies used to evaluate watershed responses to water repellency. Fire-induced water repellency has been reported in a wide range of wildland environments throughout the world, including United States, Turkey, Australia, New Zealand, Canada, and South Africa.

Water-repellent soils have an important effect on the infiltration of water into the soil and its subsequent movement and disposition throughout the soil profile. A brief description of wetting resistance to water drop penetration was presented in an earlier paper in this proceedings (DeBano 1999). The simplified contact-angle relationships of the water droplet test illustrated in this earlier paper become immensely more complicated when describing water movement through a porous media, such as a natural soil.

Fire-Induced Water Repellency

Laboratory tests of changes in water repellency resulting from different times and temperatures of heating have been combined with measured temperatures during prescribed fires and wildfires to develop the hypothesis describing how a water-repellent layer is formed beneath the soil surface during a fire (DeBano 1981; DeBano et al. 1998). According to this hypothesis, organic matter accumulates on the soil surface during intervals between fires. During these intervals, the upper soil horizons become water repellent due to the drying out of the mixture of partially decomposed organic matter and mineral soil. The addition of hydrophobic substances due to the leaching of decomposing plant parts on the soil surface may also contribute to the prefire water repellency. Fungal growth also is a dynamic source of hydrophobic substances, particularly in the organic-rich upper soil horizons.

The combination of combustion and heat transfer during wildfires produces steep temperature gradients in the surface layers of the mineral soil. During a fire, temperatures in the canopy of burning chaparral brush can reach over 1100°C (Countryman 1964). Temperatures can reach about 850°C at the soil-litter interface. But, temperatures at 5 cm in the mineral soil probably do not exceed 150°C because dry soil is a good insulator (DeBano et al. 1976). Heat produced by combustion of the litter layer on the soil surface vaporizes organic substances, which are then moved downward in the soil along the steep temperature gradients until they reach the cooler underlying soil layers, where they condense. Incipient water repellency at different soil depths could also be intensified in place by heating, because organic particles are heated to the extent that they coat and are chemically bonded to mineral soil particles. Movement of hydrophobic substances downward in the soil occurs mainly during the fire. After the fire has passed, the continued heat movement downward through the soil can re-volatilize some of the hydrophobic substances, resulting in thickening the water-repellent soil layer or fixing the hydrophobic substances in situ (Savage 1974). The final result is a water-repellent layer below and parallel to the soil surface of the burned area.

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Water Movement and Erosion

Fire affects the entry of water into a soil in two ways. First, the burned soil surface is unprotected from raindrop impact, which loosens and disperses fine soil and ash particles that can seal the soil surface. Second, soil heating during the fire produces a water-repellent layer at or near the soil surface, which impedes infiltration into the soil. If the water-repellent layer is on the soil surface, the hydrophobic particles are more susceptible to raindrop splash than if they were wettable (Terry and Shakesby 1993). The decreased infiltration produces a familiar rainfall-runoff-erosion scenario. Most important, the decreased infiltration rates increase surface runoff which initiates surface runoff and subsequent erosion. The increased surface runoff quickly entrains loose particles of soil and organic debris found on the soil surface following fire. Surface runoff may quickly concentrate into well-defined rills and increase surface erosion, particularly on steep slopes. The suspended material is then rapidly moved downslope to stream channels where it produces sediment-laden torrents that can eventually empty onto lower-lying flood plains. The effect of a water-repellent layer near, or at, the soil surface on infiltration and runoff has been easily conceptualized and has been validated in the laboratory and on small outdoor erosion plots. It has, however, been more difficult to validate on watershed-scale studies because of the large temporal and spatial variability of water repellency under field conditions. The following discussion first describes a conceptual framework of how different hydrologic processes are affected by water repellency, and then examines data collected during laboratory and field experiments.

Infiltration and Runoff

Some anomalies occur during infiltration into a water-repellent soil during laboratory studies. One such anomaly is that the uptake of water during infiltration is slower at the beginning of infiltration and increases over time, which is contrary to infiltration into a wettable soil where the converse is true (Letey et al. 1962; DeBano 1975). The reason for the increased infiltration rates over time may be due to the dissolution of water-repellent substances over time and a more complete wetting of discontinuous wettable sites on the individual soil particles over time. A second anomaly is that in water-repellent soils, faster infiltration rates occur in moist soils compared to dry soils (Gilmour 1968). This second anomaly arises because initial soil moisture content affects the initial severity of the water-repellent condition. Water repellency has been expressed as "potential" if it is measured on air-dry soil and as "actual" if measured at field moisture content (Dekker and Ritsema 1994).

The above relationship describes water flow when the soils are uniformly water repellent or wettable. However, when a soil profile contains a layer of water-repellent soil beneath a thin wettable layer (as is often the case on burned watersheds that contain a wettable ashy surface layer), the water-repellent layer affects infiltration in much the same way as a coarse-textured layer would in a wettable soil profile. If the water-repellent layer lies beneath a layer of wettable soil, the wetting front moves through the wettable layer rapidly until it reaches the water-repellent layer, after which the infiltration rate drops to that of the water-repellent soil. The infiltration rate remains depressed until the wetting front passes through the water-repellent layer into the underlying wettable soil; then the rate begins to increase (DeBano 1975). The depth to the water-repellent layer also affects infiltration rates so that a layer near the surface is more effective in restricting infiltration than a deeper layer (Mansell 1969).

The idealized model of infiltration into a soil having a uniformly distributed water-repellent layer, such as that described above, grossly oversimplifies field environments because of the large spatial heterogeneity of soil water-repellency patterns and soil surface microtopography. Studies on agricultural soils indicate that uneven microtopography of the soil surface and a heterogeneous spatial distribution of water repellency within the soil profile lead to a redistribution of surface water and concentrate water flow through the soil in discrete wettable soil fingers (Ritsema 1998). The same differential flow undoubtedly occurs in wildland soils, but under a more variable spatial environment.

Other Erosional Processes

Two important erosion processes usually occur following fire—dry ravel and rill formation. Dry ravel occurs during and immediately following the fire on steep slopes where gravity moves loose materials released during the fire downslope. Rill formation occurs when rainfall exceeds infiltration rates and causes surface runoff. Dry ravel is well illustrated in the southwestern United States where loose soil and rock material is held on steep slopes by shrubs during the interval between fires. High-severity wildfires burn the shrubs, releasing

the stored material to move downslope by gravity. Dry ravel accumulates in channels at the base of steep slopes and remains there until increased streamflow moves it downstream (Wells 1987). The greatest mover of sediment, however, appears to be rill formation accompanied by sufficient sideslope runoff to move the debris stored in the channels. Two important processes that move soil from hillslopes during rain storms are raindrop splash and rill formation.

It has been reported that raindrop splash losses for hydrophobic soils were greater than for similar wettable sandy loam soils when both were exposed to various rainfall intensities, durations, and soil surface inclinations (Terry and Shakesby 1993). Information gained through the use of synchronized videocameras has revealed that raindrop impact on hydrophobic soils produces fewer, slower-moving ejection droplets that carry more sediment to a shorter range than a wettable soil. Hydrologically, raindrop detachment is more effective on hydrophobic soils compared to wettable soils for two reasons. First, soil surfaces having an affinity for water become sealed and compacted during a rainfall event, which makes them increasingly resistant to splash detachment. Conversely, the hydrophobic soil remains dry, non-cohesive, and easily displaced by splash when the raindrop breaks the surrounding water film. Second, the splash resulting from raindrop impact differs substantially between the hydrophobic and wettable soil surfaces, as described above.

A striking feature on freshly burned watersheds during the first postfire rainstorms is the formation of an extensive rill network, which has been directly related to water repellency (Wells 1981, 1982, 1987). The sequence of rill formation follows several well-defined stages. First, the wettable soil surface layer is saturated during initial infiltration. The water infiltrates into the wettable surface until it encounters a water-repellent layer (Wells 1981). This process occurs uniformly over the landscape so that when the wetting front reaches the water-repellent layer, it can drain neither downward nor laterally. As rainfall continues, water fills all available pore space until the wettable soil layer becomes saturated. Because pores cannot drain, pore pressures build up immediately above a water-repellent layer. This increased pore pressure reduces intergranular stress among soil particles, and as a result, decreases shear strength in the soil mass and produces a failure zone at the boundary between the wettable and water-repellent layers where pore pressures are greatest. Pore pressure continues to increase and shear strength decreases until it is exceeded by the shear stress of gravity acting on the soil mass. When this happens, a failure occurs and a portion of the wettable soil begins to slide downslope. If the soil is coarse textured, initial failure causes a reorientation of the soil particles in the failure zone and they momentarily lose contact with each other. The loss of intergranular contact further reduces shear strength and extends the failure zone downslope. When most of the soil grains lose contact, a condition develops in which the shearing soil is almost fluid. This fluid condition produces a miniature debris flow in the upper wettable soil layer, which propagates down to the bottom of the slope or until it empties into a channel.

Water in the wettable soil layer adjacent to the debris flow is no longer confined and can flow out into the rill formed by the debris flow and free-flowing water runs over and erodes into the water-repellent layer. Flowing water confined to the rill still cannot infiltrate into the water-repellent soil, and therefore flows down the debris flow track as free water in an open channel (Wells 1981). As the water flows down the track, turbulent flow develops, which erodes and entrains particles from the water-repellent layer. The downward erosion of the water-repellent rill occurs until eventually the flow cuts completely through the water-repellent layer and begins infiltrating into the underlying wettable soil. Flow then diminishes, turbulence is reduced, and downcutting ceases. Finally the rill is stabilized immediately below the lower edge of the water-repellent layer. The individual rills formed by the above process develop into a network that can extend the length of a small watershed. Observations of rills after the first rainstorms on recently burned watersheds confirm that the downcutting of rills stops at the bottom of the water-repellent layer (Wells 1987).

Hillslope Plot Studies of Erosion and Runoff

Using small plots has been a popular technique for studying water repellency under field conditions. These plots have been particularly useful for studying hillslope runoff and erosion. In a study on small hillside plots under a eucalyptus forest in Australia, it was concluded that fire-induced water repellency produced localized runoff and sediment movement only on hillslopes; it did not appreciably affect the performance of the entire watershed (Prosser and Williams 1998). Another

study of plots covered with 8-year-old scrub species in Spain showed that fire intensities affected erosion, and sediment delivery was eight times greater on plots burned at high intensities than on unburned controls (Soto and Diaz-Fierros 1998). Plots have also been used to study the spatial variability of water repellency (Doerr et al. 1998) and the relationship between the spatial distribution of water repellency and the erosion potential produced during prescribed burning (Robichaud 1996).

The results of several plot studies suggest that the hydrologic responses to fire-induced water repellency depend upon soil dryness. During evaluation of the hillslope module for the Water Erosion Prediction Project (WEPP), higher runoff coefficients were consistently measured during dry periods compared to the remainder of the year (Soto and Diaz-Fierros 1998). The increased runoff was attributed to an increase in the severity of water repellency at lower soil water contents during the dry season. While studying the effect of litter applications on overland flow from small burned and unburned plots in Portugal, two mechanisms were identified as being responsible for runoff. After long dry periods, overland flow was Hortonian and was linked closely to the presence of hydrophobic soils (Walsh et al. 1994). During wet periods, however, soils lost their hydrophobicity and overland flow resulted from a perched water table developing in shallow soils. A study on small plots in Portugal also concluded that during extended dry periods, latent soil hydrophobicity appeared to become reestablished, leading to increased runoff generation and soil loss (Terry 1994).

Watershed Responses

Predicting watershed responses by using information gained from conceptual models, laboratory studies, field observations, and runoff and erosion data from small plots is extremely difficult because extrapolation of the relationships to a watershed scale often misses the increased variability found in these heterogeneous and highly complex natural systems. One useful technique for evaluating watershed responses to different treatments is to use paired watersheds, with the control and treated watersheds having been calibrated against each other for several years before and following a treatment (in this case, prescribed fire or wildfire). Reports of several studies in South Africa are used here to illustrate how watershed-level studies can be designed and the responses evaluated when studying watershed responses to fire-induced water repellency (Scott 1993, 1997; Scott and Van Wyk 1990). All of these studies involved coordinated measurements of streamflow response, sideslope erosion, and soil water repellency.

In one study, paired watersheds were used to compare streamflow and sediment responses to fire, with supplemental data on water repellency and soil loss collected on hillslopes in the calibrated watersheds. Streamflow and suspended sediment were measured on both watersheds before and after fire at permanent stream gauging stations. Soil losses were measured using 18 x 3 m plots on hillslope study sites following the fire. Water-repellency tests were taken throughout the watershed at four soil depths. The watershed sideslopes had been reforested with *Pinus radiata* to replace an indigenous cover of sclerophyllous scrubs (fynbos). The channels of both watersheds supported native riparian forests. In 1986 an intense wildfire burned about 80 percent of one watershed. During the first year following the fire, the weekly streamflow total increased 12 percent, stormflow increased 62 percent, quick flow volumes increased 201 percent, peak flow rates increased 290 percent, and the watershed response ratio increased 242 percent as a result of the fire (Scott and Van Wyk 1990). The second-year responses were somewhat less, with stormflow volume increased 20 percent, quick flow volumes increased 47 percent, peak discharges increased 108 percent, and watershed response increased 88 percent over that for the unburned watershed (Scott 1997). Soil loss by overland flow from the plots during the first year following fire increased from 10 to 26 t/ha, and both suspended sediment and bedloads increased about fourfold following the fire. Wettability of the soils was decreased significantly and the most severe water repellency was found deeper in the soil.

In a different experiment in South Africa, four treated and three control watersheds were used to evaluate the effect of prescribed fires and wildfires on watersheds with different vegetative covers (Scott 1993). Two of the treated watersheds were covered with indigenous native fynbos—one with *Pinus radiata* forest and one with *Eucalyptus fastigata* forest. One of the two fynbos watersheds was prescribe-burned and the second was burned by a wildfire during the wet season. Both of the forested watersheds were burned by an intense wildfire during the dry season. Data from runoff plots and on water repellency were collected on treated watersheds. The two forested watersheds

burned by the intense wildfires experienced significant increases in stormflows and soil loss. After fire, the storm hydrographs were higher and steeper, though their duration was little changed. The first-year increases were 190 and 1110 percent for peak discharge, 210 and 92 percent for quickflow volume, and 242 and 319 percent for the pine and eucalyptus watersheds, respectively. Neither of the fynbos watersheds showed a change in stormflow although annual flow increased 16 percent because of reductions in transpiration and interception. Water-repellency measurements suggested that the stormflow responses were partly generated by increased surface runoff into the stream channel that occurred as a result of reduced infiltration into water-repellent soils on the hillslopes.

A third study utilized a nested watershed design, supplemented with hillslope plots and water-repellency measurements (Scott and Schulze 1992). This study was designed to evaluate the effects on stormflow and hillside erosion of a high-intensity wildfire that burned a eucalyptus forest. The fire markedly increased stormflows and caused high soil losses from the hillslopes. The increased overland flow was linked to the widespread presence of water repellency. Measured soil losses of the hillslopes, however, were about five times that measured at the stream gauging stations because a healthy riparian area acted as an effective buffer, trapping large amounts of eroded soil and ash.

The most complete studies describing the linkage have been reported by David Scott in South Africa. In one study (Scott 1993) the streamflow, stormflow, and sediment yield were measured on four catchments in South Africa following a fire. Two of the catchments were covered with overmature scrub vegetation (fynbos) prior to burning, a third with eucalypt forest (*Eucalyptus fastigata*), and a fourth with pine (*Pinus radiata*). One of the fynbos catchments was prescribe-burned and all other watersheds were burned by wildfires. The catchments were instrumented to determine changes in total streamflow volume, some stormflow characteristics, and the sediment yields of each catchment in terms of suspended sediment and bedload. Soils were sampled for water repellency at 12 to 15 locations in each major vegetation type on two catchments to assess the effect of fire on soil wettability. On the remaining catchments, only brief field surveys were carried out after the fires to determine the extent of water repellency in soils; they produced only a qualitative assessment. In addition, overland flow plots (3 x 22 m) were established after the fires on two of the watersheds. On the other two watersheds plots were established but only total sediment yield was measured. The differences in burning conditions (prescribed fire versus wildfire) and the vegetation cover (scrub and forest trees) produced several measurable differences. Under severe fires, produced when heavy, dry fuel loads were consumed, postfire erodibility was increased. Prescribed burns, particularly after rains, did not completely consume fuel materials. Vegetation types that lead to the development of hydrophobic soils (eucalyptus and pine) produced sharp hydrological responses that played a part in generating surface runoff following fire. Neither of the two fynbos watersheds produced substantial increases in stormflow or total flow increases. In contrast, on the two timbered catchments, substantial increases in stormflow and soil losses occurred. The effects of fire were considered to be due to changes in stormflow generation consistent with an increased delivery of overland flow (surface runoff) to the stream channel. This was caused, in part, by the reduced infiltration resulting from water repellency in the soils of the burned catchments. Overall, the hydrological responses to fire were related to numerous interactive factors, including the degree of soil heating, the vegetation type, and the soil properties.

Summary

It has been well established that water repellency can be intensified by soil heating during a fire. It is easy to relate the effect of water repellency in soils to water movement in the laboratory or on small hillslope plots, and these responses have been confirmed by numerous well-designed experiments. Extrapolating information gained during laboratory studies and small plots, however, to entire catchments is complex because of the spatial and temporal variability of fire-induced water-repellency patterns found in the soil. Further complicating the isolation of the effects of water repellency on catchment performance is the knowledge that fire also reduces vegetative cover, destroys surface litter, degrades soil structure, and changes a host of other parameters that can affect the overall hydrologic performance of a catchment. Very few experiments have evaluated on-site water repellency, hillslope hydrology, and watershed response simultaneously.

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WATER USE BY EMORY OAK IN SOUTHEASTERN ARIZONA

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Conservation and sustainable use of oak woodlands in southeastern Arizona within the framework of integrated resource management requires more knowledge about their ecological functioning, in particular the water use of both individual oak trees and stands of oak trees. Water use by the oak trees affects the water budget and therefore the responses of other ecosystem components. Tree harvesting and regeneration, and vegetative conversion treatments, are likely to change the proportion of water available to other parts of the hydrologic cycle. There are also questions concerning how oak trees adjust their water demands and become established, survive, and grow on relatively dry sites. Understanding oak ecophysiology is critical to understanding the dynamics within these ecosystems. Knowledge about the relative water use of oak trees could influence decisions related to reforestation strategies. The initial results of a study designed to increase such knowledge are reported herein.

Study Methods

Water use by mature and sprout forms of Emory oak (*Quercus emoryi*) trees was estimated by the sapflow velocity method (see below) on two sites in the oak woodlands of the Coronado National Forest in southeastern Arizona. The two sites lie on a south-facing slope of the Huachuca Mountains in the San Rafael Valley along the international border. The dominant soil on the two sites, at 1750 m in elevation, is the Casto-Martinez-Canelo type formed from parent materials consisting of mixed sedimentary and igneous rock (Hendricks 1985).

The oak woodlands on these sites are representative of the more extensive encinal (oak) woodlands found in the U.S. Southwest and northern Mexico. Selective cutting of fuelwood last took place on one site in 1980 under permits issued by the USDA Forest Service, whereas fuelwood harvesting last occurred 100 years ago on the other site. Stocking conditions on the former are indicative of a site that had been recently harvested for fuelwood; the latter site mimicked the stocking conditions commonly encountered in unharvested oak woodlands.

Mature trees (60 years and older) with a few scattered stump sprouts and root suckers comprised the oak community on the unharvested site, where five of the mature trees were sampled. Mature trees (standards) and a few pre- and numerous post-harvesting stump sprouts characterized the stocking conditions on the harvested site, where five standard and five post-harvesting stump sprouts (a total of 10 trees) were sampled. Sapflow measurements obtained on all 15 sample trees on approximately a biweekly schedule for 18 months (from the middle of 1996 through 1997) were converted into estimates of water use (transpiration) by the sampled trees on a daily and seasonal basis. These estimates were then averaged to obtain estimates of water use for the sample of mature trees on the unharvested site, and mature trees (standards) and post-harvesting stump sprouts on the harvested site.

Sapflow Velocity Method

The sapflow velocity method of estimating water use by plants is based on measurement of the rates of flow within the water-conducting tissue of the sampled trees (Brooks et al. 1997). A heat pulse is injected into the sap stream of the trees at a point in the main stem. Subsequent measurements of this tracer are made both upstream and downstream from the source. The method assumes that the net upward velocity of sap represents an index of transpiration rate, and is useful for indicating periods when transpiration is occurring. Cross-sectional areas through which sap is flowing are

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calculated from measurements of tree diameter at breast height (dbh) outside the bark, average bark thickness of the stem, and average sapwood widths obtained (in this study) from increment cores to estimate fluxes of transpiration through time.

A sapflow meter, originally described by Swanson (1962) and Swanson and Lee (1966) and modified for this study, was used to inject the heat pulse and measure the movement of the tracer. A historical review of the theory, instrumentation, and techniques of thermal methods for measuring sapflow in trees is presented in Swanson (1994).

Initial Results and Discussion

Although Emory oak is considered a drought-deciduous species, none of the measured trees lapsed into complete dormancy in terms of sapflow meaurements. Contributing factors could be water storage in the trees, water storage in the soil, reduced competition for available water because of dieback of herbaceous vegetation, or a combination of these factors.

There were little differences in daily water use between the mature Emory oak trees on either site for most of the measurement dates (Figures 1 and 2). This finding was largely attributed to the mature trees on both sites having similar crown volumes (and therefore photosynthesis capacities), xylem volumes, and rooting systems. Possible explanations for the differences in daily water use, which occurred on about 15 percent of the measurement dates, include differences in the time of day that each tree was measured in the field and changes in meteorological factors (such as temperature, relative humidity, cloud cover, radiation intensity, and so forth) during the time that the mature trees on one site were measured relative to the time of measurements on the second site.

However, there were differences in daily water use between the mature trees and the post-harvesting stump sprouts on the harvested site, which were measured on all of the measurement dates (Figure 3). Average daily water use for the 18-month study period by a mature tree on either site was nearly 17.5 liters, whereas about 4 liters of water were used daily by individual post-harvesting stump sprouts. This result was likely due to the morphological and physiological differences between the two tree forms. Crown and xylem volumes for the stump sprouts were considerably less than those for the mature trees, although rooting systems of the two tree forms are assumed to be similar.

The relationships of daily water use by Emory oak trees and meteorological factors were also investigated. The daily water use by Emory oak trees appeared to increase after a short lag time following the cessation of a major rainfall event. This lag was thought to be due to the time it takes for rain water to infiltrate the soil and percolate to a depth where it can be accessed by tree root systems. There was little relationship between daily water use and air temperature, or between daily water use and relative humidity.

Future Analyses

Future analyses of the study data will include the formulation of regression equations relating annual water use by the sampled trees to individual tree characteristics; these regression equations will then be used to derive estimates of water use for extrapolation to a whole stand. Individual tree characteristics to be considered include diameter at the root collar (drc), dbh, total height, and crown volume. Crown length and crown volume will also be regressed to provide possible proxy values for crown volume. The regression equation for estimating annual water use as a function of either dbh or drc will serve as a basis to estimate annual water use by stands of Emory oaks. These estimates will be obtained for stands that are representative of unharvested and harvested stocking conditions through extrapolations of the regression equation to respective inventories of stocking conditions in the immediate vicinity of the sites sampled. The effects of fuelwood harvesting or other forms of tree removal with regard to water use by Emory oaks can then be analyzed.

Future analyses will facilitate development of a more comprehensive water budget for oak woodlands in southeastern Arizona. To achieve this purpose, estimates of annual water use (transpiration) by stand will be converted to a one-dimensional measurement (mm) of depth. These estimates will then be compared to annual precipitation amounts to determine what proportion of the precipitation is used by Emory oaks in the two types of stands sampled. Changes in the water budget because of fuelwood gathering or other forms of tree harvesting can be imputed from these estimates.

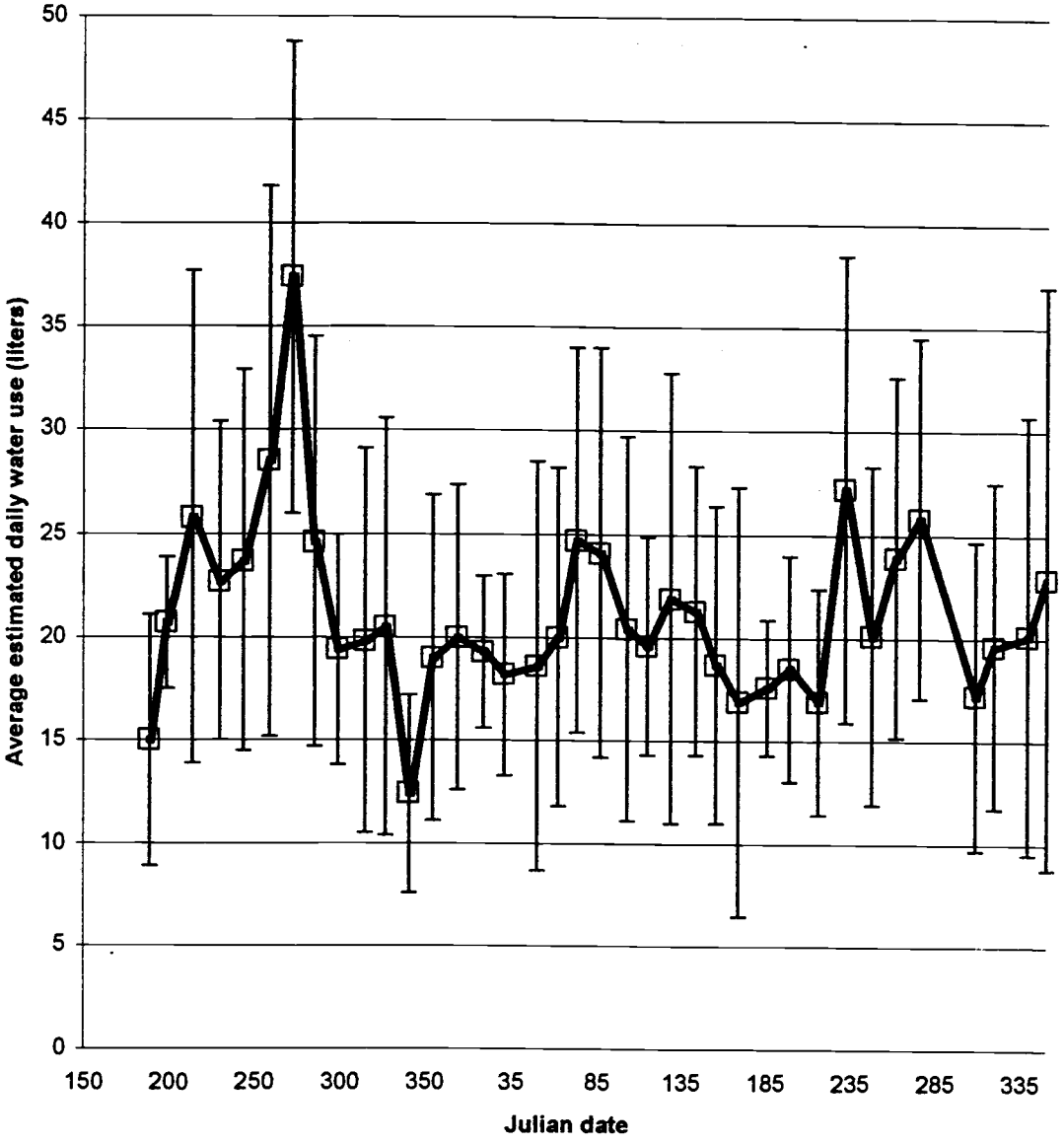

Figure 1. Average daily water use (with standard deviations) by mature Emory oak trees on the unharvested site.

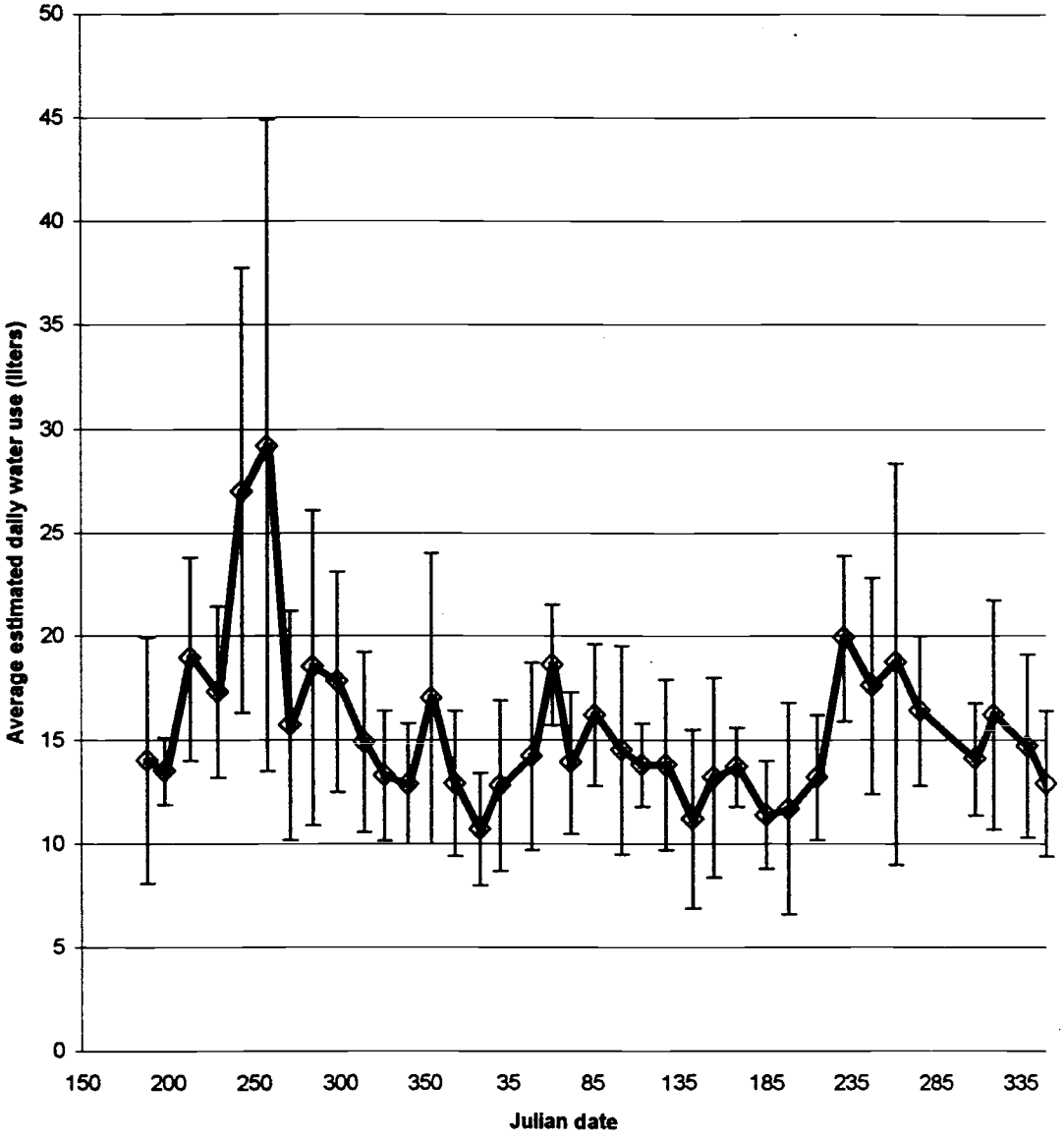

Figure 2. Average daily water use (with standard deviations) by mature Emory oak trees (standards) on the harvested site.

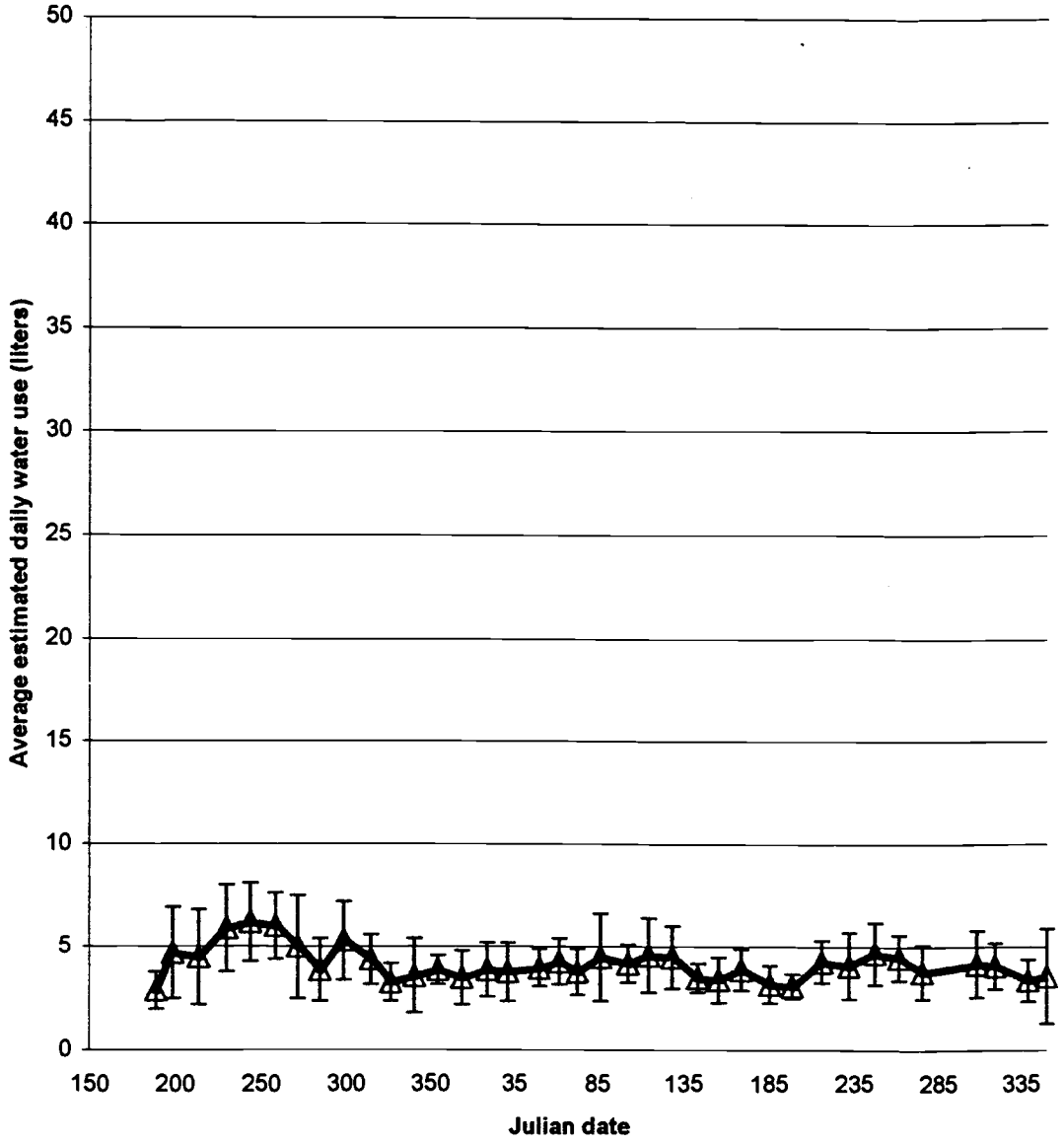

Figure 3. Average daily water use (with standard deviations) by post-harvesting Emory oak stump sprouts on the harvested site.

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IMPACTS OF VEGETATIVE TREATMENTS ON SEDIMENT CONCENTRATIONS FROM THE BEAVER CREEK WATERSHEDS IN NORTH-CENTRAL ARIZONA

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Estimating sediment production and export from watersheds is helpful in predicting on-site and off-site environmental impacts of vegetative treatments. Besides causing siltation in downstream reaches and depositions in reservoirs (Grenney and Heyse 1985), sediment in itself is a major pollutant and a carrier of plant nutrients, pesticides, and other chemicals (Glymph 1972; Osterkamp and Parker 1991; Duff et al. 1996). Although *sediment load* is an important parameter for estimating sediment buildup in reservoirs, *sediment concentration* is a primary factor of environmental concern to land managers (Wetzel 1983; Grenney and Heyse 1985).

One method of analyzing the effects of vegetative treatments on sediment concentrations is through interpretation of sediment rating curves that relate sediment concentration to streamflow discharge (Lopes and Ffolliott 1993; Lopes et al. 1996). A sediment rating curve reflects the pattern of soil erosion and sediment delivery operating on a watershed, and provides a readily accessible starting point for investigating the impacts of vegetative treatments on sediment discharge. This paper discusses the derivation of sediment rating curves to estimate impacts of vegetative treatments on sediment concentrations from the Beaver Creek watersheds in north-central Arizona. Improvement in developing these sediment rating curves by partitioning the data set into streamflow-generating mechanism and hydrograph stage is also discussed.

Study Area

The Beaver Creek watersheds, located about 80 km south of Flagstaff, are situated in the Salt and Verde River Basin of north-central Arizona. These watersheds are representative of the extensive areas of ponderosa pine forests and pinyon-juniper woodlands found in the southwestern United States. Descriptions of vegetative characteristics, physiological features, and precipitation-streamflow regimes of these watersheds have been presented by Brown et al. (1974), Clary et al. (1974), and Baker (1982) and will therefore not be detailed here.

The most important precipitation from a streamflow-generation standpoint is that originating from frontal storms during October through April, when about 60 percent of the annual precipitation falls. A second precipitation season is July through early September, when high-intensity, short-duration, localized convectional storms are common. Most annual runoff is produced from melting snowpacks in March or April. Winter runoff accounts for 85 percent of the annual water yield (Baker 1982). Suspended sediment discharges are 75–80 percent of the total sediment discharge from the watersheds studied.

Vegetative Treatments Evaluated

The vegetative treatments evaluated on ponderosa pine watersheds consisted of creating cleared openings in the forest overstories and reducing forest overstory densities. WS 12 (184 ha) was completely cleared. All merchantable timber was removed and the remaining nonmerchantable wood, and all intermingling Gambel oak and alligator juniper, were felled in 1966–67. Residual slash and debris was machine windrowed to trap and retain snow, reduce evapotranspiration losses, and increase surface drainage efficiency. The windrows were burned in 1977 to determine whether their removal would influence water yield (Baker 1983). Ponderosa pine, Gambel oak, and alligator juniper

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were allowed to seed themselves or sprout and grow following the clearing treatment. Because hydrologic changes caused by the treatment cannot be separated from those caused by the windrows, the evaluated treatment consists of complete forest clearing, soil disturbances due to timber harvesting operations, and soil disturbances due to the presence of windrows. The treatment resulted in an average increase in annual water yield of about 30 percent (44.5 cm) for 7 years after treatment, at which time the post-treatment response became insignificant.

On WS 14 (546 ha), one-third of the ponderosa pine forest was cleared in irregular strips averaging 18 m wide in 1970. Slash was piled and burned in the cut strips. The forest overstory in the intervening leave strips, which averaged 37 m wide, was reduced by thinning to about 25 percent or 18 m^2 ha^{-1} of basal area, a density level thought optimal for subsequent growth. Slash and debris were piled and burned in the cleared strips. This treatment resulted in a 57 percent reduction in basal area on the watershed. Gambel oak was retained throughout the watershed for mast and browse production, which is important to indigenous wildlife. Annual water yield increased about 20 percent (24 cm) in the first 4 post-treatment years, after which the response was insignificantly low or negative.

WS 13 (369 ha) served as a control against which the completely cleared and strip-cut treatments were evaluated. Although some commercial timber had been harvested in the early 1950s, conditions on this watershed at the time of this study represented those obtained through minimal managerial inputs.

Treatments on the pinyon-juniper watersheds consisted of converting the woodland overstories to covers of less water-consuming herbaceous plants. On WS 1 (131 ha), a cabling treatment was applied in 1963. Larger trees were uprooted by a heavy cable pulled between two bulldozers. Smaller trees missed by cabling were hand-chopped, slash was burned, and the watershed was seeded with a mixture of forage species. This treatment did not result in significant increases in annual water yields.

On WS 3 (147 ha), a mixture of picloram (2.8 kg ha^{-1}) and 2,4-D (5.6 kg ha^{-1}) was applied by helicopter to 114 ha in 1968; the remaining 33 ha were either not treated or the trees were sprayed with a backpack mist-blower. This treatment resulted in a significant increase in annual water yields of about 160 percent (11.4 mm) for 8 post-treatment years, after which the residual dead trees were removed in a firewood sale.

WS 2 (51 ha) was a control against which the cabling and herbicide treatments were evaluated. Conditions on this watershed represented those obtained through minimal managerial inputs.

Acquisition of Data Sets

Suspended sediment concentration and streamflow data obtained from 1974 through 1982 were the source data for this study. Data sets reflecting immediate impacts of the vegetative treatments were excluded from the analysis to better describe long-term impacts of the treatments on sediment concentrations. Either grab samples, integrated samples obtained with a DH-48 hand sampler, or pumping samples were analyzed by filtration to determine sediment concentrations. Streamflow was measured in concrete trapezoidal flumes. When a sample of suspended sediment was collected, the time was indicated on a digital tape on the continuous water-level recorders at the gauging stations. Sediment data were collected for streamflow discharges in excess of 0.05 m^3 s^{-1} and at time intervals greater than 1 hr to reduce possible effects of serial correlation.

Three types of events served as the basis for studying the effects of streamflow-generation mechanisms on sediment concentrations:

Type 1: Snowmelt-runoff events not preceded by precipitation; relatively slow response time to peak streamflow discharge; streamflow duration of several days or weeks; occurs in late winter–early spring.

Type 2: Mostly convectional rainfall events of high intensity and short duration; rapid response time to peak streamflow discharge; streamflow duration of hours or a few days; occurs in late summer–early fall.

Type 3: Essentially frontal rainfall events of low intensity and relatively long duration, mostly in late fall and winter months; insignificant snow accumulations on the ground; moderate response time to peak streamflow discharge.

Event types 1 and 3 generated most of the streamflow events studied in the ponderosa pine forests, whereas event types 1 and 2 resulted in streamflow events studied in the pinyon-juniper woodlands. Rain-on-snow events, which were major events when they occurred, represented less than 10 percent of the individual stream-

flow-generation mechanisms on the watersheds studied and were therefore excluded from this analysis.

Derivation of Sediment Rating Curves

Sediment rating curves are frequently expressed in terms of a power function form, such as $C = aQ^b$, where C = suspended sediment concentration (mg/L), Q = streamflow discharge (m^3/s), and a, b = regression coefficients for a particular stream. Sediment rating curves to be derived in a power function form are often approximated by least-square linear regressions of logarithmic-transformed data (Walling 1977); this transformation was used in this study. The procedure to overcome the possibility of bias when the regression estimates were detransformed and to minimize spurious conditions has been outlined by Duan (1983).

The coefficient of determination, R^2, was used to compare goodness-of-fits of the derived sediment rating curves. An adjusted coefficient of determination, Ra^2, was used to correct for the dependence of goodness-of-fit on degrees of freedom:

$$Ra^2 = R^2 - \frac{P(1-R^2)}{N-P-1}$$

where N = number of observations and P = number of independent variables = 1.

Analysis of the derived sediment rating curves began with the complete data sets from each of the watersheds studied. These data sets were subsequently partitioned into the types of streamflow-generation mechanism (snowmelt runoff or convectional rainfall) and then into the hydrograph stage. Suspended sediment samples collected in periods of high streamflow discharges were assigned the same weight as measurements made during low streamflow discharges in deriving the sediment rating curves. Parameters a and b of the sediment rating curves for the ponderosa pine watersheds, with the 95 percent confidence limits, fitted standard errors, coefficients of determination, and F statistics, are presented in Table 1. Similar statistics for the pinyon-juniper watersheds are shown in Table 2.

Comparative Analysis and Discussion

A hierarchical test was made to determine the statistical significance of changes in explained variations of sediment concentrations, expressed in terms of R^2, at the watershed, streamflow generation, and hydrograph stage levels. The null hypothesis at each level was that the parameters a and b of two sediment rating curves had not changed when the data sets were disaggregated. This null hypothesis was formulated as

$$H_0 : a_1 = a_2 = \ldots = a_m, b_1 = b_2 = \ldots = b_m$$

where m = number of equations tested. At each level, the Chow test (Kmenta 1986) was performed in a pairwise fashion to test the null hypothesis that the parameters (a and b) of two sediment rating curves had not changed significantly (at the 95% level of significance) due to partitioning. Test results are shown in Tables 1 and 2, and the comparative analysis and the accompanying discussion are presented below.

Ponderosa Pine Watersheds

There were significant differences in sediment rating curves among the treated ponderosa pine watersheds and control watershed at the watershed level. These differences indicated that for a similar streamflow discharge, sediment concentrations from the completely cleared watershed (WS 12) were significantly higher than those from the strip-cut watershed (WS 14), and that sediment concentrations from the strip-cut watershed were higher than those from the control watershed.

WS 12 experienced watershed-wide soil disturbance from the complete clearing treatment, simultaneous breaking up of the herbaceous ground cover by the clearing operation, and follow-up pushing of the residual slash and debris into windrows. Soil disturbance on WS 14 was less extensive and more localized than what took place on WS 12. The most destructive portion of the disturbance on WS 14 occurred on the third of the watershed that was cut into irregular strips, where much of the protective herbaceous plant cover was disturbed, and where the residual slash and debris were piled and burned. Larger snowpack accumulations occurred in the strip-cuts in comparison to those in intervening leave strips. These strip-cuts had up-down slope orientations, causing increased overland water flows originating from the larger snowpack buildups concentrated in the strips, where most of the sediment production on WS 14 took place.

Streamflow-generation mechanisms on the ponderosa pine watersheds were snowmelt runoff (type 1) and frontal rainfall (type 3). Although sediment rating curves derived at the

Table 1. Sediment rating curve parameters, with 95% confidence limits, standard errors, coefficients of determination, and F statistics for ponderosa pine watersheds.

Watershed	Event Type	N	a	95% Confidence Intervals		b	95% Confidence Intervals		Standard Error SE	r_a^2	F Statistics	Significance
12	all	353	154.579	121.667	196.413	1.042	0.947	1.137	125.97	0.57	468.54	**
	1	326	155.234	120.370	200.224	1.066	0.964	1.167	125.39	0.57	426.87	**
	1r	148	170.847	114.559	254.700	1.027	0.877	1.177	169.38	0.55	182.82	**
	1f	131	141.415	99.093	201.393	1.198	1.058	1.399	64.91	0.69	282.32	**
	3	26	162.577	83.813	315.726	0.865	0.634	1.097	135.65	0.70	59.38	**
	3r	12	175.942	46.664	663.722	0.880	0.693	1.267	182.39	0.69	25.69	**
	3f	14	127.938	14.280	14.280	0.746	0.287	1.205	98.82	0.47	12.55	**
13	all	204	28.679	21.804	37.717	0.677	0.579	0.776	61.40	0.47	183.40	**
	1	179	26.272	19.909	34.678	0.665	0.560	0.770	64.44	0.47	156.44	**
	1r	44	39.853	15.288	103.837	1.030	0.745	1.315	94.37	0.55	53.35	**
	1f	71	21.670	12.754	36.782	0.608	0.425	0.790	56.86	0.38	44.01	**
	3	25										N/S
	3r	8	76.552	10.800	121.456	0.537	0.512	1.164	24.14	0.46	6.94	**
	3f	12	36.196	44.945	70.906	0.938	0.886	1.062	73.24	0.74	32.77	**
14	all	473	56.590	43.335	68.681	0.974	0.919	1.097	74.94	0.50	475.87	**
	1	432	74.788	75.944	112.071	1.008	1.019	1.294	97.63	0.53	494.69	**
	1r	140	44.905	30.817	65.431	1.156	0.740	1.018	46.72	0.66	276.26	**
	1f	154	80.902	42.763	153.134	0.879	0.314	0.815	50.12	0.50	156.45	**
	3	41	112.158	54.425	231.101	0.578	0.245	0.746	55.15	0.37	24.31	**
	3r	20										N/S
	3f	9										N/S

N = sample size
F statistics = equation significant at 0.05
Significance = regression significance (**) at $\alpha = 0.05$.
all = sediment rating curve using all measurements
1, 2, 3 = event types 1, 2, 3, respectively
r = rising stage of event hydrograph
f = falling stage of event hydrograph
N/S = not significant

Table 2. Sediment rating curve parameters, with 95% confidence limits, standard errors, coefficients of determination, and F statistics for pinyon-juniper watersheds.

Watershed	Event Type	N	a	95% Confidence Intervals		b	95% Confidence Intervals		Standard Error SE	r_a^2	F Statistics	Significance
1	all	525	7.362	5.297	10.257	0.233	0.143	0.324	7.03	0.05	39.33	**
	1	429	6.982	4.831	10.069	0.216	0.114	0.318	7.37	0.04	17.26	**
	1r	93	66.222	26.122	105.925	0.779	0.632	0.956	9.79	0.54	11.01	**
	1f	333										N/S
	2	90	6.668	3.013	14.757	0.231	0.025	0.438	4.17	0.04	4.94	**
	2r	20	38.905	10.116	149.624	0.529	0.190	0.868	2.92	0.34	10.77	**
	2f	70										N/S
2	all	519	5.129	3.664	7.161	0.193	0.112	0.274	2.98	0.04	22.01	**
	1	448	4.335	3.013	6.237	0.168	0.081	0.256	3.13	0.03	14.24	**
	1r	153	13.772	6.730	28.119	0.438	0.253	0.625	4.23	0.12	21.82	**
	1f	288										N/S
	2	65	14.125	6.124	32.509	0.363	0.157	0.568	2.74	0.15	12.37	**
	2r	12	16.672	10.789	25.763	0.172	0.073	0.270	1.22	0.51	15.16	**
	2f	53	29.923			0.613				0.46	45.88	**
3	all	611	8.091	5.916	11.066	0.245	0.168	0.321	4.67	0.06	25.76	**
	1	572	8.147	5.984	11.092	0.244	0.168	0.320	4.95	0.06	40.13	**
	1r	203	29.174	17.179	49.431	0.547	0.408	0.685	6.66	0.23	60.46	**
	1f	362	4.457	3.097	6.412	0.120	0.034	0.206	2.31	0.02	7.48	**
	2	33										N/S
	2r	9										N/S
	2f	23										N/S

N = sample size
F statistics = equation significant at 0.05
Significance = regression significance (**) at $\alpha = 0.05$.
all = sediment rating curve using all measurements
1, 2, 3 = event types 1, 2, 3, respectively
r = rising stage of event hydrograph
f = falling stage of event hydrograph
N/S = not significant

watershed level were not significantly different from those derived for frontal rainfall events for WS 13, they were different for WS 12 and WS 14. This result was not surprising, as the process of dislodging and transporting soil particles by low-intensity, relatively long duration, frontal rainfall events (type 3) would be less effective on watersheds with more vegetative cover (WS 13). It follows that estimates of sediment concentrations from sediment rating curves for the ponderosa pine watersheds would be improved by separating sediment data derived from snowmelt runoff from those derived from the complete data sets.

Hydrographs were partitioned into rising and falling stages to evaluate the commonly observed statement that the rising stage of a hydrograph is generally associated with higher rates of suspended sediment transport than the falling stage (Elliott and DeFeyter 1986; Glysson 1987; Brooks et al. 1997). The results indicated a greater difference between sediment rating curves derived after the data sets were partitioned into rising and falling stage hydrographs. Sediment concentrations for the rising stage of a hydrograph were generally higher than those for the falling stage. This finding reinforced the previous findings that higher rates of sediment transport during stormflow events are found with the rising limb of a hydrograph.

Pinyon-Juniper Watersheds

There were differences in sediment rating curves among the treated pinyon-juniper watersheds and the control watershed at the watershed level (Lopes et al. 1996). The main difference was higher sediment concentrations from the cabled watershed (WS 1) than the control watershed for similar streamflow discharges. Higher concentrations of suspended sediment on WS 1 were likely a reflection of the soil disturbances caused by uprooting trees in the cabling treatment.

There was also a difference between sediment rating curves derived for the watershed treated with herbicides (WS 3), which experienced little soil disturbance as a result of treatment, and the control watershed. However, soil disturbance caused by the follow-up removal of merchantable firewood, and piling and burning the residual slash 8 years after the herbicide treatment, was apparently significant in terms of affecting suspended sediment discharge.

There were significant differences in sediment rating curves derived from the total data sets and the curves derived from the data sets after they had been partitioned by streamflow-generation mechanisms for WS 1 and WS 2. However, there was no significant difference in sediment concentration when streamflow-generation mechanisms were considered on WS 3. Similar to the situation observed on the ponderosa pine watersheds, a large portion (85%) of the total data sets was associated with snowmelt runoff events; therefore, these latter events dominated the statistical properties.

Sediment rating curves derived from data sets partitioned into rising and falling stage hydrographs were different from those derived at the streamflow-generation levels of analysis for convectional rainfall events (type 2) on WS 2 and snowmelt-runoff events (type 1) on WS 3. There were no differences in suspended sediment concentration for the rising and falling limbs of the hydrograph for the other cases. Therefore, with the exception of hydrographs generated by convectional rainfall events (type 2) on WS 2 and by snowmelt-runoff events (type 1) on WS 3, results from the pinyon-juniper watersheds do not support the statement that higher rates of sediment transport are found with the rising limb of a hydrograph.

Conclusions

Soil disturbances caused by vegetative treatments on Beaver Creek watersheds in both vegetative types generally increased sediment concentrations significantly above those of control watersheds. This response is reflected by their respective sediment rating curves. Completely cleared and strip-cut ponderosa pine watersheds produced higher suspended sediment concentrations than did the control watershed. Likewise, cabled and herbicide-treated pinyon-juniper watersheds yielded higher sediment-laden streamflows than did the control. It is important to emphasize, however, that the elevated suspended sediment concentrations observed following the treatments in both of the vegetative types were still low relative to water-quality standards.

Although more than 85 percent of the data analyzed represented snowmelt-runoff events in both vegetative types, derivation of sediment rating curves based on streamflow-generation

mechanism improved the sensitivity of the analysis. This improvement was particularly true for watersheds that had higher levels of disturbance.

Sediment concentrations were generally higher in the rising stage than in the falling stage for ponderosa pine watersheds. However, there was no clear evidence of higher sediment concentrations in the rising stage of the hydrograph as compared to the falling stage in the pinyon-juniper watersheds.

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THE STATUS OF SPIKEDACE IN THE VERDE RIVER, 1999: IMPLICATIONS FOR MANAGEMENT AND RESEARCH

John N. Rinne

The threatened spikedace, *Meda fulgida*, was once widespread and locally abundant in streams and rivers of the Gila River Basin. It is a federally listed threatened species, occurring in four isolated stream and river systems in Arizona and New Mexico. Based on 6 years of study of the Verde River population, the spikedace has declined dramatically in abundance, even to the point of non-detection in samples since 1997. Calculated probabilities suggest between 8 and 9 chances out of 10 that the species is extirpated from the upper Verde River. Based on current data, river hydrograph and nonnative fish play interactive roles in this decline. Complicating determination of relationships and causal factors has been removal of livestock grazing from the Verde River riparian stream corridor and the resulting changes in riparian stream habitat.

From an intrastream perspective, monitoring for spikedace on the upper Verde River should be intensified. In the event of reappearance, local experimental suppression or reduction of nonnative fishes in these reaches should be considered to observe the spikedace and native fish community response. Also, refugia streams should be identified and evaluated, culture techniques should be developed, and priorities for research established. One suggested research opportunity is comparative studies with the upper Gila River, New Mexico, to refine our understanding of the relationships and interactions of flow regimes, native-nonnative fish abundance, and land-use activities.

Verde River Spikedace Population

The upper Verde River (upstream of Sycamore Creek) is a free-flowing stream with low baseflows (0.57 m^3/sec), only one perennial tributary (Granite Creek), and periodic extreme flood events (Stefferud and Rinne 1995). Although adapted to the vagaries of southwestern stream dynamics, over the past 6 years spikedace populations have appeared to be affected dramatically by natural variations in the Verde River hydrograph (i.e. alternating floods and drought; Stefferud and Rinne 1995; Rinne and Stefferud 1997). In addition, introduced nonnative fishes such as red shiner (*Cyprinella lutrensis*; Rinne 1991; Douglas et al. 1994) and larger predatory species such as smallmouth bass (*Micropterus dolomieui*), green sunfish (*Lepomis cyanellus*), and yellow bullhead (*Ictalurus natalis*) appear to have negatively affected the species (Rinne 1991; Stefferud and Rinne 1995) through a proportional increase in the total fish community.

In addition, livestock grazing has been suggested to negatively impact spikedace habitats and abundance. Notwithstanding, no reports or published accounts are available that document the nature and extent of this impact in the upper Verde River or elsewhere in southwestern riparian stream systems. The only information implicating this land-use activity as negatively affecting spikedace habitat and populations was a draft biological opinion (administratively withdrawn) on one grazing allotment on the upper Verde River. In general, information is lacking on the relationships between native southwestern cypriniform fishes and livestock grazing (Rinne 1999).

The primary objectives of this paper are threefold: (a) Examine the current status of the spikedace based on temporal-spatial distribution and abundance in the upper Verde River. (b) Evaluate and discuss factors that may have affected the current status of the species in the upper Verde River. (c) Make short- and long-term recommendations for management and research that may aid in both sustaining and enhancing this threatened species of native minnow in the upper Verde River.

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Description and Biology

The spikedace is a slender, laterally compressed cyprinid. The dorsal surface is typically olive gray to brown, often mottled, the sides are usually silvery, and the ventral region whitened. Males become golden on their dorsal and lateral surfaces during breeding. The basic life history of the species has been studied primarily in one stream in Arizona—Aravaipa Creek (Barber and Minckley 1966, 1983; Barber et al. 1970; Minckley 1981; Turner and Tafanelli 1983; Rinne and Kroeger 1988). In addition, Anderson (1978), Propst et al. (1986), and Propst and Bestgen (1986) have studied the biology of the species in the Gila-Cliff reach of the mainstream Gila River in New Mexico. However, most of the information collected on the species has been from survey and monitoring activities to assess the distribution and abundance of the species in time and space (LaBounty and Minckley 1973; Anderson and Turner 1977; Barrett et al. 1985; Propst et al. 1986) and to provide limited information on the biology of the species.

Spikedace spawn in the spring and early summer (April to June), and are principally dependent upon streamflow and attendant water temperature, normally varying in time and space. In early March, 1999, extreme coloration of males and gravid females were noted in the Gila-Cliff valley of New Mexico (personal field observations). Eggs are expelled in the water column and are adhesive when fertilized, adhering to the substrates where spawning occurs. Sand to gravel-pebble substrates have been reported where reproductively ready individuals (i.e. based on coloration and gravidity) have been collected in the upper Verde River (Neary et al. 1996), Aravaipa Creek (Barber et al. 1970), and the upper Gila River, New Mexico (Propst et al. 1986).

Historic Distribution and Current Status of Spikedace

The spikedace, *Meda fulgida*, is a diminutive, short-lived, stream-dwelling minnow endemic to the Gila River Basin of Arizona and New Mexico (Minckley 1973; Miller and Hubbs 1960). Although once widespread in the Gila River Basin (Rhode 1980; Propst et al. 1986), the spikedace is currently federally listed as a threatened species (U.S. Fish and Wildlife Service 1986) and presently occurs only at intermediate elevations (1070–1830 m) of the upper Gila River in southwestern New Mexico (Propst et al. 1986; U.S. Fish and Wildlife Service 1990), Aravaipa Creek (Barber and Minckley 1966; Bettasco et al. 1995), Eagle Creek (W. L. Minckley, unpublished data), and the upper Verde River (Stefferud and Rinne 1995).

Historically, the spikedace was probably widespread and locally abundant in the Gila River Basin from low to mid-elevation reaches of the San Pedro (type locality, Miller and Hubbs 1960), Salt, Verde, Agua Fria, San Francisco, and Gila rivers. Although present at upper elevations (1525–1830 m) of these same mainstream rivers, population numbers were likely lower. Gaps in quantified temporal-spatial information such as museum collections do not permit unequivocal assessment and delineation of historic distribution. Generally, the species was common throughout the basin and likely locally abundant in preferred habitats. As with many western native cyprinids, fluctuations in both range and numbers of spikedace in response to regional environmental and climatic conditions are the norm (Minckley 1973). Another related, rare native cyprinid, the Little Colorado spinedace (*Lepidomeda vitatta*), ranges from abundant to rare in habitats of its native range and is extirpated from certain local stream systems (Minckley and Carufel 1967). For both species, extrinsic and intrinsic factors may likely be delimiting variations in range and abundance; however, specific factors have not been adequately defined.

Recent Distribution and Abundance in the Upper Verde River

Although historically present in the Verde River, the spikedace was not re-discovered in samples until the early 1970s from the mouth of Sycamore Creek (Anonymous 1974). Large numbers of spikedace were collected by the U.S. Bureau of Reclamation in the mid 1980s in the upper Verde while examining instream flow needs of native fishes (U.S. Fish and Wildlife Service 1989). Spikedace were common in samples and comprised up to 11 percent of the native fish fauna.

The Rocky Mountain Research Station, Flagstaff Arizona, commenced study of fish populations in the upper Verde in 1994 following major flooding (75-year recurrence event) in the winter of 1992–93 (Stefferud and Rinne 1995). Spikedace were most abundant in the most upstream reaches of the upper Verde in 1994 (Table 1; Stefferud and Rinne 1995). Fewer were collected downstream at the Perkinsville and Black Bridge sites and at the mouth of Sycamore Creek. A reduced level of flooding (10-year recurrence event) occurred again in March 1995. Spikedace populations pulsed once

Table 1. Fish community composition at seven sampling sites in the upper Verde River, 1994–1999.

	1994	1995	1996	1997	1998	1999
NATIVE SPECIES						
Longfin dace	1319	12	282	21	13	2
Desert sucker	2644	328	471	231	126	167
Sonora sucker	1810	322	654	240	125	118
Roundtail chub	776	341	259	50	64	25
Spikedace	428	72	140	0	0	0
Speckled dace	171	25	68	1	12	2
NONNATIVE SPECIES						
Yellow bullhead	31	29	9	40	33	15
Common carp	23	6	13	19	9	4
Red shiner	1473	97	275	2238	1047	545
Channel catfish	5	2	0	1	0	0
Mosquito fish	0	0	0	3	6	59
Flathead catfish	0	1	1	1	1	0
Green sunfish	4	29	6	8	21	49
Smallmouth bass	14	10	32	35	66	104
Fathead minnow	7	0	0	0	0	0
Total fishes	8750	1274	2210	2288	1523	1090
Percent native	82	86	85	19	22	29

more in 1996 (Rinne and Stefferud 1997), only to decline in subsequent sampling to complete absence from samples at the seven established sites by spring 1997 (Rinne 1998). Additional sampling along the entire course of the upper Verde in 1997 and 1998 also indicated that the spikedace population had declined markedly. In the spring of 1999, sampling in the most upstream reaches of the Verde (Bear Siding upstream) again indicated no spikedace (unpublished data). Based on these data, the species presently is very rare in the upper Verde.

To assess and address more specifically the rarity and probability of extirpation of spikedace from the upper Verde, the approach of Grogan and Boreman (1998) was applied. This methodology uses years and last year of collection versus total years of surveys to arrive at a probability that a species, based on historic collection, is extirpated. A probability (P) of 0.873 that spikedace are gone from the Verde was calculated using U.S. Forest Service data since 1994 and a last date of collection of 1997. A more robust data set from Arizona State University that commences with collections in 1980 and has a last date of collection of 1997 resulted in a lower, but similar probability (P = 0.832) of extirpation of this species from the upper Verde River, above Sycamore Creek.

Habitat

Spikedace occupy lentic habitats of varying depths (<1 m) over gravel and pebble substrates. The species is often found in greater abundance in shear zones where two riffle areas converge to form eddying currents (Rinne 1985, 1991, 1992). In larger rivers, the species is most common in riffle areas of moderate velocities (30–50 cm/sec) and gradient (0.5–1.0%; Neary et al. 1996; Rinne unpublished data) over pebble-cobble and boulder substrates. The latter substrate frequently provides alteration or reduction of current velocity and creates eddying currents that appear to be one of the species' preferred habitats. Habitat associations may vary in both time and space as well as ontogenically (Anderson 1978; Rinne 1985; Propst et al. 1986; Propst and Bestgen 1986; Rinne and Kroeger 1988; Rinne 1991).

River Hydrograph

A study of the fish community of the upper Verde River relative to abiotic and biotic factors was conducted in 1997–98 by the U.S. Forest Service, Rocky Mountain Station with Arizona Heritage funding (Rinne 1998). The primary objective of the study was to determine the relative influence of the hydrograph and introduced fishes in delimiting the relative native to nonnative composition of the total fish community. The results of the 3 years of study (Rinne and Stefferud 1997) show that native fishes respond positively to flooding. Further, Rinne (1998) reported that the hydrograph and the lack of flooding over the entire Verde River have enhanced nonnative over native fishes. By spring 1998, after 3 years (1996–98) of no significant (i.e. greater than bankfull) flow events, the relative abundance of native fishes (>70%) was reduced to the extent that nonnative fishes comprised this same relative percentage of the total fish community. Rinne (1998) concluded that flooding was essential for sustaining a native fish fauna in a river system such as the upper Verde with nonnative fishes present.

Spikedace populations paralleled the overall population trend of native fishes in the upper Verde River (Rinne et al. 1998). The species was common in samples at four of the seven established sampling sites in the spring of 1994, rare by the spring of 1996, and absent from samples in spring of 1997 (Table 1). Therefore, flow regimes appear to be important to essential for sustaining populations of spikedace and other native species, especially in the presence of nonnative fishes.

Nonnative Fishes

Although Rinne and Stefferud (1997) concluded after 3 years of sampling that the river hydrograph was more influential than nonnative fishes for sustaining a native fish component in the upper Verde, it appears that the interaction of the two factors combine to legislate the relative native to nonnative composition of the overall fish community. That is, the lack of any significant flooding and sustained base flow since 1996 has resulted in nonnative fishes increasing significantly to comprise the majority (>70%) of the fish community (Table 1). Accordingly, in the absence of significant flow events, nonnative fish species have become dominant. Only with the advent of flooding of at least the 1995 level (5+ year recurrence) can this hypothesized relationship of hydrograph and nonnative fish abundance be more completely defined.

In summary, if our studies had been conducted only during 1994–1996, flooding could be readily defended as the controlling factor in delimiting spikedace populations in the upper Verde. By contrast, studies during only 1997–1999 could lead to the conclusion that nonnative fishes are the controlling factor in delimiting spikedace numbers in the upper Verde. These data emphasize the importance of continual monitoring up to and following the next flood event (10-year recurrence or greater).

Grazing Management Changes, Verde River Corridor

Changes in grazing management on the river corridor are confounding the question of which factor (i.e. flooding or nonnative fishes) is most influential in determining the status of all native fishes, and specifically spikedace. Since the spring of 1997, and following continuous reductions in animal unit months (AUMs) over the previous decade, livestock grazing has been totally eliminated from the Verde River corridor. As a result, riparian vegetation diversity, density, and biomass have increased dramatically on the river corridor (Al Medina, unpublished data; Figure 1). Neither native fishes nor spikedace have so far responded positively to this change in management (Figure 1), although nonnative fishes have paralleled the increases in vegetation both on streambanks and within aquatic habitats. These relationships or correlations need additional study.

Conversely, there has been an inverse relationship between decreases in native fish populations, livestock numbers, and flooding, and corresponding increases in nonnative fishes (Figure 1). There are clearly several factors interacting here (i.e. fishes, livestock, flooding, and vegetation). Lack of flooding has apparently been favorable to vegetation and nonnative fishes and unfavorable to native fishes, including the spikedace. The dramatic increase in vegetation since 1997 due to the removal of grazing parallels an increase in nonnative fish abundance. The marked increase in two predatory nonnatives, smallmouth bass and green sunfish (Table 1), may be a response, in part, to greater instream cover associated with increased nearstream and streambank vegetation density. These correlations and relationships could be coincidental; they have not yet been adequately studied or statistically tested.

Discussion

Notwithstanding known cycles of abundance (Minckley 1981; Propst et al. 1986), the current status of the spikedace in the upper Verde River is perilous at best. They have declined in just a few years from 6 or 7 percent (1994 and 1996, respectively) of the native fish community to a level of non-detection. This decline coincided with an increase in base flow over the past 2 decades (Neary and Rinne 1997), but also with a period of no flooding. The combination of the predominance of nonnative fishes, the probability of entering a drought cycle, and the change in riverine habitat resulting from grazing management changes could lead to extirpation of one of only two large river populations of spikedace in the Southwest. Although statistically nonsignificant, the high probability that spikedace is extirpated from the upper Verde is cause for concern.

Of these three factors, flooding, nonnative fishes, and habitat change, only two can be addressed by management. Flood events are subject to the vagaries of the southwestern climate. Alternating cycles of drought and flood are the rule, not the exception. Data from the upper Verde River (Stefferud and Rinne 1995; Rinne and Stefferud 1997; Brouder in review) and the upper Gila River (Propst et al. 1986; Rinne et al. in press), and the conclusions of other documented effects of flooding on fishes (Minckley and Meffe 1987), show that native fish populations, including spikedace, appear to be positively correlated to flooding. In the upper Verde River, spikedace have gone from common to very rare to absent from samples in just a few years during a period with no flood events and a drought or base flow condition (Rinne and Stefferud 1997; Rinne et al. 1998).

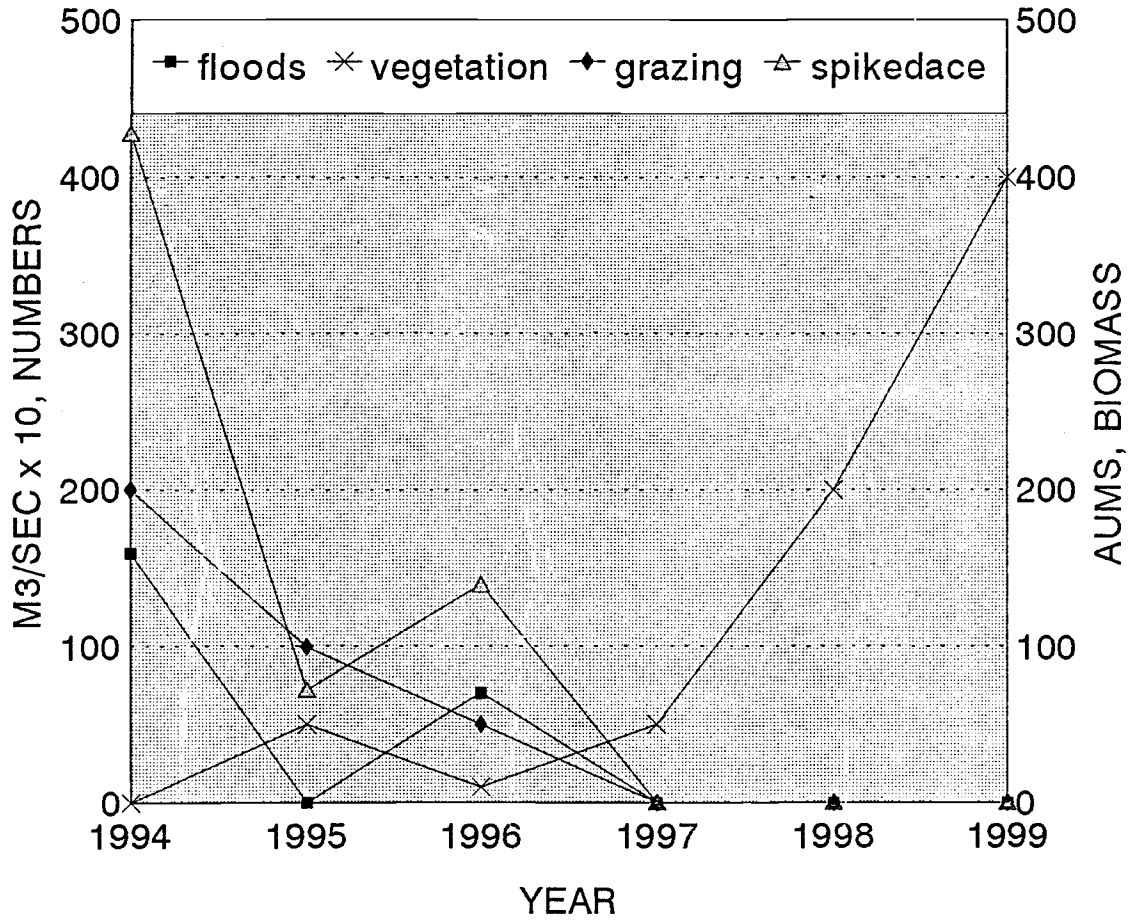

Figure 1. General trends in relationships of floods (m^3/sec x 10), grazing (AUM trend), riparian vegetation (biomass trend), and spikedace (numbers) in the upper Verde River, 1994–1999.

Of equal importance in the upper Verde, as elsewhere in the West (Minckley and Deacon 1991) and Southwest (Rinne 1994, 1997), is that nonnative fishes have consistently demonstrated a negative impact on native fishes. Whether by displacement or replacement (Douglas et al. 1994), nonnatives apparently out-compete and prey upon native species (Minckley 1983; Meffe 1985; Rinne 1995; Blinn et al. 1993). Predation may indeed be the primary factor of replacement of native fish species by nonnatives (Marsh in press). At present, more than 70 of every 100 individual fish inhabiting the upper Verde River are nonnative. Combined with the lack of collection of spikedace for the past 2 years, this is cause for both concern and focused management by state and federal agencies.

Finally, livestock grazing is a manageable land-use activity. Anecdotally and coincidentally, livestock removal from the river in 1997 has paralleled both the lack of flooding and the decline of spikedace in the upper river (Figure 1). The indirect impact of livestock grazing on the watershed of the upper Verde presents a possible threat to spikedace sustainability. Although not discountable, the relative impact of livestock grazing on the watershed, when compared to the predominance of a competitive and predatory nonnative fish fauna in the upper Verde River, becomes a moot point in the near term for management alternatives.

Complete removal of nonnative fishes from the upper Verde, which is also a highly improbable near-term management alternative, should

very likely be the highest priority for management at present. Although immediate, complete removal of nonnative fishes is not administratively or logistically feasible, efforts to suppress or reduce their abundance should be considered as a timely management alternative. Waiting for the next flood event to reduce nonnatives as a management alternative is not an appropriate management strategy because of the time factor, the current status of the spikedace, and the predominance of nonnative fishes in the upper Verde River. However, synchronizing future nonnative fish suppression with natural reduction in numbers by flooding (Rinne and Stefferud 1997) may have merit as a viable management alternative.

Flooding is a natural disturbance that may indeed be a prerequisite for native fish sustainability in the presence of nonnative fishes. Historically, floods and droughts have caused extreme fluctuations in the numbers and ranges of native fish, including the spikedace. After reduction of the entire fish community, recolonization from refugia reaches (Sedell et al. 1990) has been possible historically in the absence of nonnative fishes and without many of the current mainstream dams and their alteration of flow regimes (Rinne 1994, 1995) and removal of fluvial connectivity.

The upper Verde has no significant mainstream impoundments or diversions, only a nonnative fish assemblage. Further, with recent low drought flows and the removal of livestock, the riparian stream ecosystem of the upper Verde has become stabilized for the near term. Such stable aquatic conditions appear to be favorable to nonnative fishes in this river system. Without nonnatives, the stability characterized by adequate and perhaps enhanced baseflow (Neary and Rinne 1997) and the increase in instream and streambank vegetation, even in the absence of floods, would probably be favorable to native fishes. Nevertheless, until the next significant peak flow event occurs, nonnative fishes will continue to impact the spikedace—an impact that could result in extirpation of this threatened species from the Verde River system.

Conclusions and Management Recommendations

Currently, the spikedace is markedly reduced in population size in the upper Verde. Calculated probabilities suggest between 8 and 9 chances out of 10 that the species is gone from the upper Verde River. Even if a residual population exists, and even if spawning does occur annually, the probability of survival of young-of-year spikedace is low given the excessive numbers of nonnative, predatory species present in the river (Stefferud and Rinne 1995, Table 1). One appropriate management alternative might be to immediately begin suppression of nonnative fish populations in the most upstream reaches of the upper Verde. Such action could help sustain the spikedace locally (if present in future surveys) until the next flood event. Removal of grazing from the watershed or limits on sport fishes are commendable management strategies; however, both are cosmetic at best, and pale in significance to mechanical reduction in abundance of nonnative fishes, especially when coordinated with natural hydrologic reduction. Even mechanical, partial suppression of nonnative fishes is conceivably ineffective, unless instituted immediately following a significant flood event and a concomitant reduction in the total fish community (Rinne and Stefferud 1997).

Because of genetic considerations, the introduction of spikedace from other river systems cannot currently be considered a viable management alternative (Tibbets and Dowling 1996). Isolation of the spikedace in the upper Verde River from those of the Gila River and Aravaipa Creek dates at least to completion of the Horseshoe and Bartlett dams on the Verde in the 1930s. Analyses of spikedace populations in the Gila River system (Tibbets and Dowling 1996) suggest that the population of spikedace in the upper Verde is genetically distinct. Accordingly, no translocation of spikedace should occur unless there is unequivocal evidence that the spikedace has been extirpated from the upper Verde River drainage. At what point do we reach an unequivocal conclusion—at extirpation? Despite extensive sampling in Eagle Creek, Arizona, spikedace were not collected for more than 2 decades, but the species was recently confirmed with great effort at a disturbed road-crossing site in this stream. However, none of these situations justify adopting a no-action, non-progressive, or pro-active approach to management.

Three major additional management strategies should be strongly considered and instituted immediately. The first is intensification of survey and monitoring activity to search for spikedace. Second is locating and evaluating possible refugia streams in the Verde Basin. Last, culture techniques must be developed immediately for the species. These last two items should be in place when the next flood event occurs, which is presumably when spikedace will reappear in samples

and become abundant enough to facilitate transfer to a hatchery for propagation or stocking into identified refugia streams.

A study of the role of disturbances in native fish ecology, and spikedace specifically, also should be initiated. Comparisons should be made of successional responses of spikedace in the upper Verde with other spikedace (and native fish) populations. Following significant flooding of the Gila River resulting from Hurricane Linda in September of 1997, spikedace are currently (spring of 1999) one of the most common species of fish in the upper Gila River system (37% of the native fish community; Rinne et al. in press). A few spikedace were collected in spring-summer of 1998 in this same river. Further, nonnative fish are scarce in the upper Gila. Of more than 6000 individual fish collected between March and July of 1999 in this river system, only 10 percent were nonnatives (Rinne et al. in press).

The opportunity therefore exists to examine the population dynamics of spikedace and several other Gila River native species, such as longfin dace (*Agosia chrysogaster*), desert sucker (*Catostomus clarki*), Sonora sucker (*Catostomus insignis*), roundtail chub (*Gila robusta*), and loach minnow (*Rhinichthys* [*Tiaroga*] *cobitis*), relative to the hydrograph and in the absence of or reduction in nonnative fishes. A parallel study on the upper Verde could be conducted that examines the response of native fishes relative to mechanical reduction or suppression of nonnative fish abundance. There is a window of opportunity to examine flooding disturbance and fish community structure and succession from two extant scenarios: the abundance of natives and especially spikedace combined with a few nonnatives in the Gila River and Aravaipa Creek; and the abundance of nonnatives, the presence of a few natives, and an apparent lack of spikedace in the Verde River.

Finally, the study effort should be a coordinated regional approach to understanding the problem of floods and fish community composition, not that of a single river system or jurisdiction. There is an excellent opportunity for collaboration among Forest Service management and researchers and state agencies to understand the interrelationships of flooding, disturbance, and native-nonnative fish community structure and succession in the Southwest. The rarity of spikedace in the upper Verde River, its abundance in the upper Gila River, the abundance of nonnative fishes in the former river and scarcity in the latter, and the short-lived nature of the spikedace, are all urgent reasons for a coordinated regional management-research approach to address sustaining the spikedace in southwestern streams and rivers.

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