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Introduction

After a hiatus of 3 years, we are pleased to bring back into circulation an important document for hydrologists in the Southwest. The impetus for this undertaking began last year and we hope there is enough demand for this publication that its continuance will be assured.

Because of the 3-year break, for indexing purposes this issue is Volume 22–25. Any literature citation should also reflect this nomenclature.

The meeting that took place in Flagstaff, Arizona on April 22, 1995 was jointly sponsored by the Arizona Section of the American Water Resources Association and the Hydrology Section of the Arizona–Nevada Academy of Sciences. This publication was underwritten by Northern Arizona University's Graduate College, which also provided editorial assistance. We gratefully acknowledge the support we have received and especially want to thank Dr. Henry O. Hooper and Ms. Louella Holter, Bilby Research Center, for their assistance.

Malchus B. Baker, Jr.
Charles C. Avery

EVALUATION OF WATER BALANCE MODELS: AN ASSESSMENT IN MIXED CONIFER FORESTS OF ARIZONA

Peter F. Ffolliott[1] *and Gerald J. Gottfried*[2]

Much of the surface water used in Arizona originates as precipitation, often snow, on the higher elevation watersheds. However, the surface water supplies from these watersheds are generally limited in most years, although it has been shown that multiple-resource management practices implemented on higher elevation watersheds can favorably affect subsequent streamflow and water quality characteristics (Rich 1972; Rich and Thompson 1974; Baker 1986; Gottfried 1991; Gottfried and Ffolliott 1992). A capability to predict the streamflow from higher elevation watersheds before and after the implementation of multiple-resource management practices using water balance models based on easily acquired information is needed. Such predictions would be helpful to managers in planning for the possibility of increased streamflow into the river systems of Arizona and, as a consequence, the availability of more water for downstream users.

One major objective of watershed research in Arizona has been, and continues to be, the validation of water balance models and, where appropriate, making modification of these models to better represent the measured parameters: annual streamflow totals, timing of streamflow regimes, and such. Results from previous investigations by Baker and Carder (1977), Baker and Rogers (1983), and Jeton (1990) show that many of the existing water balance models require some modifications—input variables, functions, coefficients—to simulate the actual conditions that have been measured.

This paper reports on a study to evaluate the accuracy of three water balance models in simulating annual streamflow totals from a watershed in the mixed conifer forests of Arizona.

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Comparisons have been made between observed and simulated streamflow before and after the implementation of a treatment designed to demonstrate and evaluate multiple-resource management practices in mixed conifer forests (Gottfried 1991). Future investigations will focus largely on analytical comparisons between observed and simulated hydrographs for the same watershed.

Study Area

The south fork of Thomas Creek, within the Apache–Sitgreaves National Forest in eastern Arizona, was the study area. This 562-acre watershed is situated in the headwater region of the Salt River. It is a main tributary of the Gila River and a main source of surface water for the Phoenix metropolitan area. The watershed is located between 8,400 and 9,200 feet in elevation. Soils are classified as Mollic Eutroboralfs and Mollic Cryoboralfs, and are derived from basaltic parent materials. Annual precipitation averages about 30 inches, with approximately 55 percent of this precipitation occurring as snowfall during the October through May winter precipitation period. Annual streamflow was nearly 3.2 inches prior to the treatment. About 80 percent of the annual streamflow total occurs during the snowmelt period of March, April, and May.

The watershed originally supported undisturbed, multistoried, old-growth stands of mixed conifer species (Gottfried 1991; Gottfried and Ffolliott 1992). These stands consisted of Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), white fir (*Abies concolor*), corkbark fir (*A. lasiocarpa* var. *arizonica*), Engelmann spruce (*Picea engelmannii*), blue spruce (*P. pungens*), ponderosa pine (*Pinus ponderosa*), southwestern white pine (*P. strobiformis*), and quaking aspen (*Populus tremuloides*). The pretreatment basal area of these stands was about 185 square feet per acre.

The Treatment

Following a calibration period, the upper 75 percent of the watershed was harvested in 1978 according to single-tree selection, group selection, and small patch-cut prescriptions; the lower part of the watershed remained untreated, however, because of the steep slopes encountered. The timber harvest implemented resulted in a nearly 35 percent reduction in the basal area on the treated areas and the creation of 63 small openings representing about 13 percent of the watershed area. The treatment resulted in a significant increase in annual streamflow of almost 1.7 inches, an increment representing approximately 45 percent of the pretreatment streamflow (Gottfried 1991). The increases in streamflow were attributed largely to reduced evapotranspiration from the forest overstory and increased snow accumulations in the small openings.

Analysis

The results presented in this paper represent an analysis of the relative accuracies of the three water balance models in simulating annual streamflow totals from the south fork of Thomas Creek. Inputs to the models represented the conditions on the watershed at the initiation of the simulation exercises. In this initial phase of the study, outputs from the models, that is, the simulated annual streamflow totals, were compared to the corresponding streamflow records by a discrepancy ratio (DR) between the simulated and observed streamflow. The closer the DR value was to 1, the closer the simulated streamflow to the observed streamflow, and therefore the more accurate the model in simulating annual streamflow totals from the watershed. DR values less than 1 indicate that the simulated streamflow was less than the observed streamflow; in other words, the model being evaluated underestimated the observed streamflow. DR vales greater than 1 represented the reverse situation. The three water balance models evaluated were:

1. *The combined WATBAL–SNOWMELT simulation model*: WATBAL simulates the total water balance on an annual basis and compiles the results obtained from individual response units into a composite overview of a watershed; it emphasizes snow accumulation and melt processes (Leaf and Brink 1973). The original WATBAL simulation model assumed a continuous snowpack, a major constraint when applied in the southwestern United States, where snowpacks are generally intermittent. A modified snow component called SNOWMELT provides for modeling intermittent snowpack conditions within the WATBAL framework (Solomon et al. 1976).

2. *The Baker-Kovner streamflow regression model*: This nonlinear regression model predicts annual streamflow totals from inputs of precipitation, potential insolation, basal area of the forest overstory, and two interaction terms (precipitation–basal area and precipitation–potential insolation). The model was developed with data from the Beaver Creek watershed (Brown at al. 1974), and as a consequence is strictly applicable to watersheds with volcanic soils and climate similar to Beaver Creek. Testing of the usefulness of the model for other watershed conditions appeared justified in the opinion of the authors of this paper.

3. *The YIELD II simulation model*: YIELD II, a modification of the earlier YIELD simulation model, has two components: one that simulates streamflow in the winter and throughout the snowmelt season in the spring; and one that predicts streamflow from summer rainfall events (Ffolliott and Guertin 1988). A degree-day technique for estimating snowmelt is used in the simulation. Knowing the daily snowmelt and snowmelt runoff efficiency, that is, the ratio of total snowmelt on a watershed to the total amount of water released in the form of streamflow, allows simulation of the volume of water that leaves a watershed as streamflow. The second simulation component is largely a modification of the model developed by Rogerson (1976) to describe the hydrologic behavior of watersheds with snow-free conditions.

Simulations were made for 7 years that represent two periods: pretreatment conditions and post-treatment conditions. The four pretreatment years included 1 year of above-average (1973), 1 year of near-average (1975), and 2 years of below-average winter precipitation (1974 and 1977). The 3 post-treatment years represented 1 year of above-average (1985), 1 year of near-average (1980), and 1 year of below-average winter precipitation (1981).

Results and Discussion

Comparisons of observed and simulated annual streamflow totals and the corresponding DR

values obtained for the WATBAL–SNOWMELT simulation model, the Baker-Kovner streamflow regression model, and the YIELD II simulation model are presented in Tables 1, 2, and 3, respectively.

The combined WATBAL–SNOWMELT simulation model generally overestimated the observed annual streamflow totals. The relative magnitude of the overestimations is seemingly unrelated to the amount of winter precipitation, which produces most of the streamflow from the watershed. That WATBAL–SNOWMELT overestimated most of the observed streamflow was not surprising, because Baker and Roger (1983) reported similar findings when applying this water balance model on the south fork of Thomas Creek in an earlier study. In fact, the average overestimation for the pretreatment conditions shown in Table 1 agrees closely with that reported by Baker and Rogers, whose study consisted largely of modeling the monthly streamflow for the pretreatment period of 1966 to 1970, rather than annual streamflow totals for later pretreatment periods. These authors also divided the south fork of Thomas Creek into four response units for simulation purposes, while only two response units were delineated in the current study. The average overestimation of WATBAL–SNOWMELT for post-treatment conditions was less than 10 percent. Regardless of whether pretreatment or post-treatment conditions are considered, however, it is felt that the problems that Baker and Roger identified with WATBAL–SNOWMELT also contributed to the overestimations found in this present study. For example, WATBAL–SNOWMELT melts snow too rapidly in the winter, particularly at the end of the melt period, which likely contributes to the overestimation of streamflow.

The accuracy of the Baker-Kovner streamflow regression model in predicting the observed annual streamflow totals on the south fork of Thomas Creek appears related to the amount of winter precipitation in most years of the study. Considering pretreatment and post-treatment periods collectively, as there were few meaningful differences in the average DR values between the two periods, a close agreement was found between the observed and simulated streamflow in the years of near-average winter precipitation (1975 and 1980). However, the model underestimated the streamflow in the years of above-average winter precipitation (1973 and 1985), while it overestimated the annual streamflow

Table 1. Observed and simulated annual streamflow totals, and the corresponding DR values for the combined WATBAL–SNOWMELT simulation model (in inches).

Year	Observed	Simulated	DR
Pre-treatment conditions			
1973	14.25	19.66	1.38
1974	0.12	1.32	11.00
1975	5.52	8.52	1.54
1977	0.56	0.55	0.98
Mean	5.11	7.51	1.47
Post-treatment conditions			
1980	8.53	9.68	1.13
1981	1.50	2.09	1.39
1985	11.65	11.96	1.03
Mean	7.23	7.91	1.09

Table 2. Observed and simulated annual streamflow totals, and the corresponding DR values for the Baker-Kovner streamflow regression model (in inches).

Year	Observed	Simulated	DR
Pre-treatment conditions			
1973	14.25	10.43	0.73
1974	0.12	0.33	2.75
1975	5.52	5.50	1.00
1977	0.56	1.01	1.80
Mean	5.11	4.32	0.85
Post-treatment conditions			
1980	8.53	8.23	0.96
1981	1.50	1.32	0.88
1985	11.65	8.65	0.74
Mean	7.23	6.07	0.84

Table 3. Observed and simulated annual streamflow totals, and the corresponding DR values for the YIELD II simulation model (in inches).

Year	Observed	Simulated	DR
Pre-treatment conditions			
1973	14.25	9.44	0.66
1974	0.12	0.11	0.92
1975	5.52	3.35	0.61
1977	0.56	0.28	0.50
Mean	5.11	3.30	0.65
Post-treatment conditions			
1980	8.53	7.49	0.88
1981	1.50	1.32	0.88
1985	11.65	6.22	0.53
Mean	7.23	5.01	0.69

totals in 2 (1974 and 1977) of the 3 years when the winter precipitation was below average. Whether the results obtained with the Baker-Kovner model are reflective of the actual conditions encountered on the south fork of Thomas Creek or an aberration of the data is unknown. The relative contributions of the respective terms, and particularly the interaction terms in the regression formulation of the model, are difficult to evaluate individually because of the structure of the model.

The YIELD II underestimated the observed annual streamflow totals on the south fork of Thomas Creek in all of the years evaluated. Of the three water balance models evaluated, YIELD II also produced the most consistent results in terms of the DR values obtained, especially in the post-treatment period. The reasons for the consistent underestimations can only be speculated upon at this time. Due to the large proportion of the annual streamflow occurring during the snowmelt period in the winter and spring, the problem probably lies within the component of YIELD II that simulates streamflow in these two periods. The baseline temperature at which snowmelt is initiated in the degree-day technique for estimating snowmelt might be too low. Therefore, adjustments in this baseline temperature may be needed to "allow" a greater volume of snowmelt to take place. Additional study of the snowmelt processes in the mixed conifer forests of Arizona is required before making this adjustment, however. Snowmelt runoff efficiency, the other primary input to the model, represents a measured parameter of the conditions being simulated, and should probably not be modified in the routing procedure for simulating the volume of water that leaves the watershed.

Modifications are likely needed in all of the water balance models evaluated to more accurately and consistently simulate the annual streamflow totals from the mixed conifer forests represented on the south fork of Thomas Creek. The combined WATBAL–SNOWMELT model still appears to be unable to adequately simulate the intermittent snowpack conditions found in the southwestern U.S. In reference to the Baker-Kovner streamflow regression model, the average basal areas and precipitation amounts on the south fork of Thomas Creek are both higher than those found on Beaver Creek. It is likely, therefore, that a wider range of input data is needed to derive a set of regression coefficients that better simulates the conditions encountered on the south fork of Thomas Creek. The component of YIELD II that simulates streamflow in the winter and throughout the snowmelt season in the spring needs to be further evaluated to determine whether the degree-day technique for estimating snowmelt and the procedure for routing snowmelt water off of a watershed are appropriate to the conditions being simulated. It is anticipated that future comparisons between the observed and simulated hydrographs can provide insight on the nature of these modification in the water balance models.

Conclusions

Water balance models that are sensitive to climatic variability and the impacts of watershed management activities are necessary to plan and implement efficient water management practices. However, this and related investigations have shown that the existing water balance models generally require modifications in their formulations to simulate streamflow regimes before they become suitable for operational use—in this case, within the mixed conifer forests of Arizona. Such a conclusion should be expected, because the water balance models studied are largely deterministic in structure, and therefore must be modified before they are applied to local conditions. Nevertheless, the evaluations of water balance models presented in this and earlier papers should be helpful to managers in judging the relative value of the models is simulating particular hydrologic situations. Furthermore, these evaluations are useful in deciding which of the models, if any, warrant further testing and refinement.

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HYDRAULIC-CONDUCTIVITY MEASUREMENTS OF REATTACHMENT BARS ON THE COLORADO RIVER

William D. Petroutson, Jeffery B. Bennett, Roderic A. Parnell, and Abraham E. Springer[1]

Construction of Glen Canyon Dam and subsequent hydroelectric power generation has altered the flow regime on the Colorado River, changing physical and chemical characteristics of the downstream flow. The Bureau of Reclamation is the principal agency empowered with preparing an Environmental Impact Statement (EIS) of the impacts of Glen Canyon Dam on downstream resources in Glen and Grand canyons. Interim flow criteria were implemented during the EIS preparation period as required by the Grand Canyon Protection Act of 1992.

Recent work has determined that return current channels along the reattachment bars have become the highest areas of productivity for organic matter of all bar environments as a result of accumulation of organic matter and finer grained sediments during the interim flows (Stevens 1991). Geochemical analysis of groundwater within reattachment bars has indicated that progressive nutrient storage is produced in return-channel environments (Parnell and Bennett 1994). Flow modeling coupled with solute transport modeling is required to model cycling of nutrients between reattachment bar groundwater and the river. Determination of hydraulic conductivity is a necessary variable in the calculation of groundwater velocities used in flow modeling. The pneumatic slug test was determined to be the best method to measure hydraulic conductivity of Colorado River reattachment bars.

Purpose and Objectives

The purpose of this study was to determine the flow and transport parameters of the deposits that comprise reattachment bars on the Colorado River. The primary objective of the study was to determine the hydraulic conductivity of reattachment bar deposits. A secondary objective was to assess the spatial variability of hydraulic conductivity within reattachment bar deposits.

Study Area

The reattachment bars used in this study are located at river miles 43, 72, and 194 relative to Lee's Ferry on the Colorado River in Glen Canyon and Grand Canyon. Reattachment bars form as the river current leaves a constriction of the channel formed by a debris fan from a side canyon flood. The main current separates from the channel at a location referred to as the separation point and reattaches at the reattachment point (Figure 1A). This causes the formation of a recirculation zone with a return current along the channel boundary. Within the recirculation zone a reattachment bar deposit forms with a return-current channel incised along the cliff side of the bar (Figure 1B).

Well Installation

Fifty-eight wells were drilled on transects both perpendicular and parallel to the river on three separate reattachment bars. The wells were installed by pumping river water into a 3/4-inch diameter jetting tool inserted into a 2-inch diameter drive casing. The tool was capped in such a manner that water was forced up and to the side, carrying cuttings through the casing. The casings were driven to a depth of 10 feet for the shallow wells at each cluster, and 20 feet for the deep wells. Wells were assembled using 1.25-inch diameter schedule 40 PVC casing; each well had a 0.0025-cm slot screen 1.5 feet above the total depth of the jetted hole. The well pipe was inserted inside the 2-inch diameter drive casing. The 2-inch diameter drive casing was then withdrawn, allowing the sediment to collapse around the 1.25-inch well casing.

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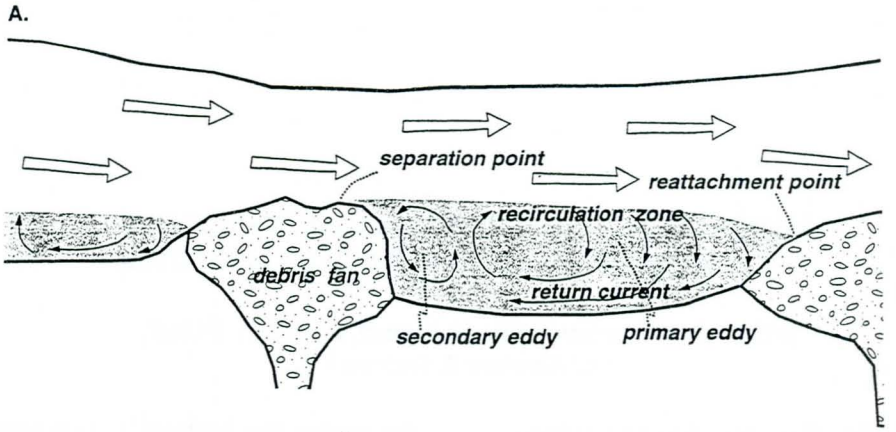

Figure 1A. Recirculation zone currents.

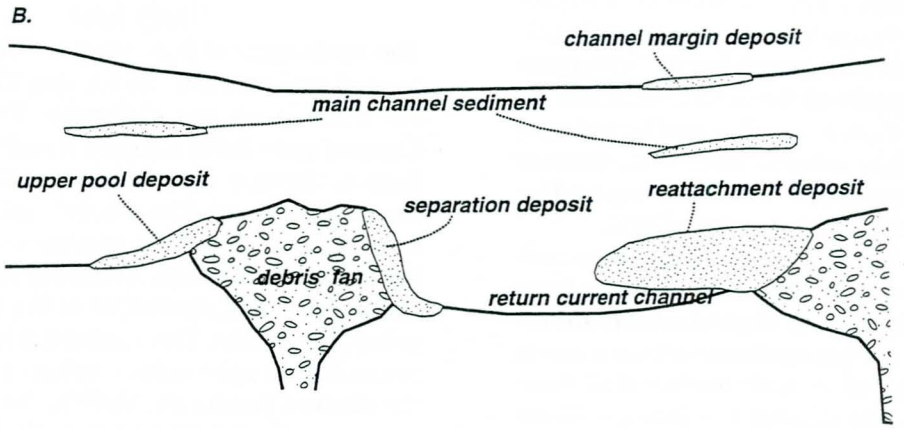

Figure 1B. Recirculation zone deposits (after Schmidt and Graf 1991).

Slug Tests

The slug test is a timely and cost effective method of obtaining a reliable estimate of aquifer parameters. However, in highly transmissive aquifers, data acquisition with a mechanical slug test is impractical because of the rapid recovery of the water level after slug removal. An alternative to the mechanical slug test is the pneumatic slug test. Detailed analysis shows that there is no statistical difference between estimates of hydraulic conductivity between these two methods (Levy and Pannell 1991).

The pneumatic slug test is conducted using a pneumatic wellhead, which consists of a 4-inch diameter PVC pipe with ports for air injection, an air gauge, and transducer emplacement. In addition, the wellhead is equipped with a 4-inch ball valve to release air pressure. The wellhead is equipped with a reducer and metal clamps to attach the wellhead to the well pipe. Data are recorded using a pressure transducer and a 2-channel datalogger. Use of the datalogger allows for the rapid acquisition of data in a short time.

The pneumatic slug test uses air to lower the elevation head of water in a well, replacing it with a pressure head. The air can be compressed, or it can be supplied by a simple bicycle pump. Head change is read off the air gauge, which is calibrated in inches of water. After a new static water level has been reached, the datalogger is initiated and the air pressure is released by turning the ball valve. The head change within the well is measured with the pressure transducer and recorded by the data-

logger, producing a head change versus time curve (McLane et al. 1990). Three tests were conducted at each well in order to assure statistical validity of hydraulic conductivity values at each site.

Slug Test Analysis

All slug tests were analyzed using the Bouwer and Rice method for unconfined aquifers (Bouwer and Rice 1976). The Bouwer and Rice method uses an empirical equation relating the effective radius over which head difference between the static water level of the aquifer and the well is dissipated, to the geometry of the well and the aquifer. The solution is achieved by fitting a line to the straight line portion of the drawdown versus time curve (Figure 2). The value of y is then substituted into the Bouwer and Rice equation, yielding a value for hydraulic conductivity. The Bouwer and Rice solution had particular utility for this study because the wells were partially penetrating, with a 1.5-foot long screen in an aquifer with approximately 40 feet of saturated thickness.

Because hundreds of aquifer tests were conducted, solutions were accomplished using an automated well analysis program. Because there were two well geometries, shallow and deep, the only variables changing between wells were static water level and initial head difference. This enabled the solutions for hundreds of trials to be obtained in hours, rather than days.

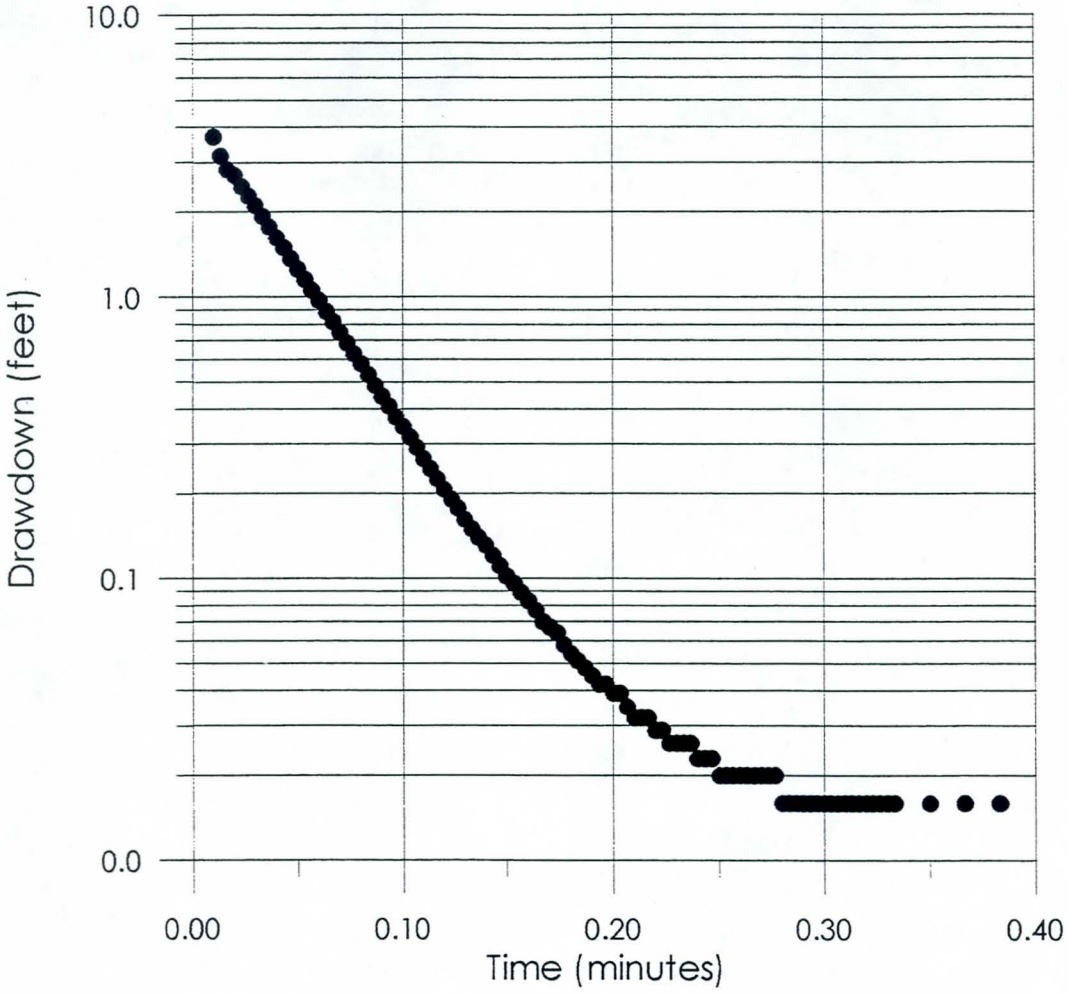

Figure 2. Time versus drawdown curve.

Summary

The values of hydraulic conductivity obtained in the study indicated a spatial variability across the reattachment bars. The largest values of conductivity were generally located at wells in the beach face. Lower conductivities were obtained for wells in the mid-beach and return-current channels. This corresponds roughly to the Schmidt and Graf model for eddy driven deposition of sand bars. In their model, the coarsest sediments would be located in the beach face where the currents are highest; these are the locations of the highest hydraulic conductivities. The lowest hydraulic conductivity values are located in the return-current channel wells where the eddy current was the slowest when the sediments were deposited.

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THE EFFECT OF DEWATERING A STREAM ON ITS RIPARIAN SYSTEM: A CASE STUDY FROM NORTHERN ARIZONA

Peter G. Rowlands, Heidemarie G. Johnson, Charles C. Avery and Nancy J. Brian[1]

The Walnut Canyon drainage system is located east of Flagstaff, Arizona (Figure 1). It includes both Upper and Lower Lake Mary reservoirs for the city of Flagstaff, Arizona, and forms the steep-sided canyons within the Walnut Canyon National Monument. Although Walnut Creek was probably a permanently flowing stream in the distant past (early Holocene and late Pleistocene), recent historical evidence indicates that it was an ephemeral stream even prior to the construction of Lower Lake Mary dam in 1904, subject only to seasonal flooding. Since the construction of the Upper Lake Mary dam in 1941, streamflow in Walnut Canyon is limited to rare events requiring a combination of snowmelt and rainfall, which result in large quantities of water entering the catch basins of both Upper and Lower Lake Mary and subsequent overflow from these two reservoirs.

Discharge has never been measured in the Walnut Canyon drainage: Only anecdotal descriptions or photographs of high-flow events are recorded. There are no gaged records of flow available for Walnut Canyon proper. However, Blee (1988) conducted a study to determine evaporation and seepage losses at Upper Lake Mary. Between 1969 and 1971 he used a continuous stage recorder to collect lake stage data at the upstream and downstream ends of the lake. The lower lake gage was permanent and the upper lake gage was temporary, operated only during May through October. The study showed that evaporation losses were 27 percent and seepage losses were 45 percent.

Weekly water records for Upper Lake Mary have been collected from 1950 to 1993 by the city of Flagstaff. This information consists of 2242 weekly records in the form of lake surface elevation, volume of water in the lake, and withdrawals for municipal use. These data form the sole basis for determination of streamflow changes in Walnut Canyon since the construction of the dams and impoundment of water.

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Historical Data

Information about the prehistoric water flow in Walnut Canyon is limited. However, freshwater mollusks in stream deposits in Walnut Canyon, near Winona, have been radiocarbon dated at 11,000 years BP. Based on stratigraphic evidence, Reger and Batchelder (1971) concluded that these mollusks inhabited a slow-moving, perennial meandering stream. Underlying deposits indicate that prior to this time, the creek was a wide, shallow channel with a flow regime capable of moving gravel. The 11,000 year date coincides with the last ice age (early Holocene) and glaciation of the San Francisco Mountain.

Historical accounts from 1853–1854, 1857–1858, and 1870, as well as an historical photograph dated 1886 (Figure 2), within the area now encompassed by the National Monument, suggest that flows in Walnut Canyon have been ephemeral at least since the 1850s, occurring seasonally with snow melt and summer rain. Streamflow may also have been ephemeral in the past when the pre-Columbian Sinagua Indians inhabited the area (500–1130 A.D.). Prior to 1905, water from summer rains and winter snows flowed down Walnut Canyon for a few weeks each year. After 1905, water only entered the canyon when Lower Lake Mary overflowed.

The Walnut Canyon watershed was dammed in 1900. The earthen dam, begun in 1904 and completed in 1905 (Coconino Sun, 1926), was 309 m long, 11.4 m high, and 29 m wide, and the reservoir first filled in March 1905. In November 1940 an additional earthen dam, measuring

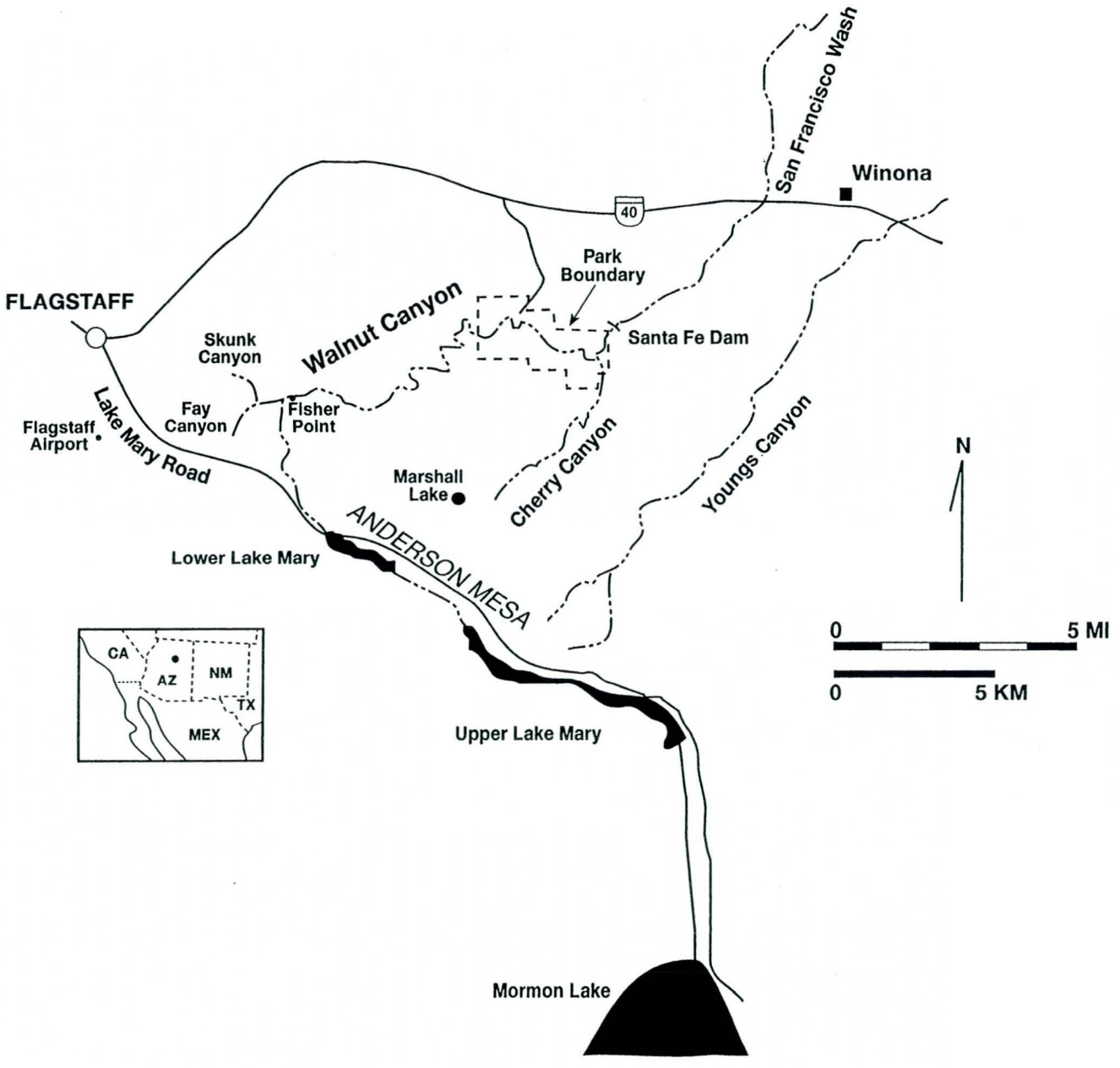
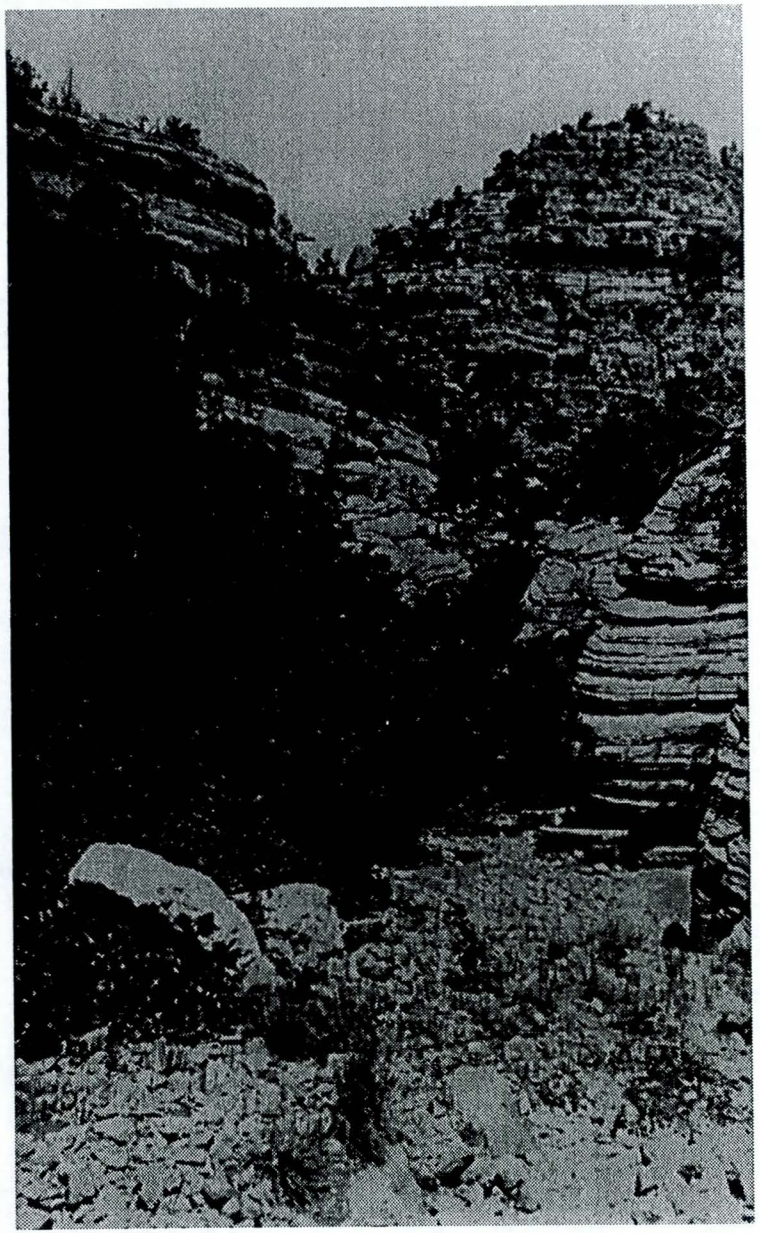


Figure 1. Map of the study area.



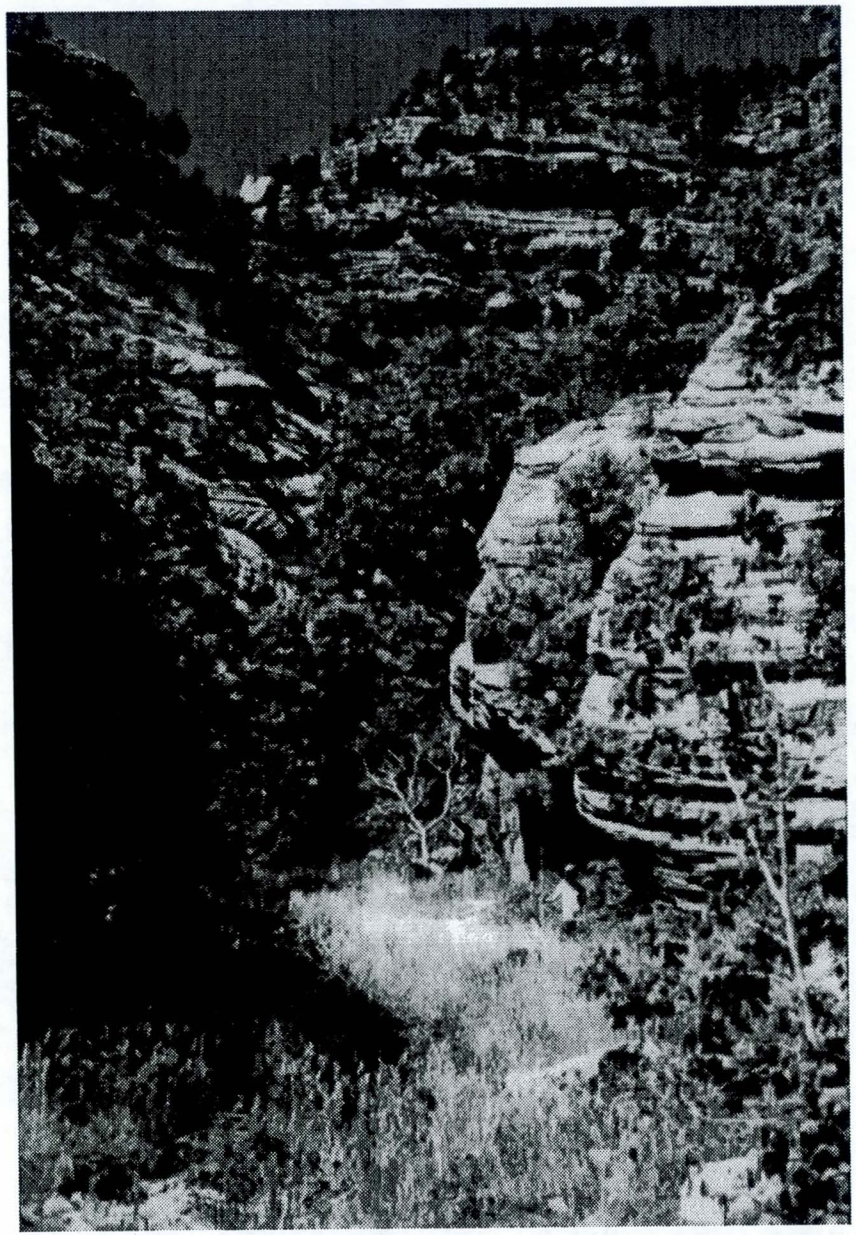


Figure 2. Photographic comparisons of Walnut Canyon, bottom east of Third Island Fort. (Left) Dry canyon bottom at Walnut Canyon, dated 1886 (Edgar A. Mearns collection, photo no. 121, labeled "Cosnino Canyon," from the Library of Congress). (Right) Rephoto of the same site taken July 1994 by Peter G. Rowlands.

267 m long, 11.1 m high, 4.3 m wide at the top, and 49 m wide at the bottom, was constructed in the drainage, 6 km above Lower Lake Mary reservoir. The spillway level was 7.95 m above the lake bottom. Completed in July 1941 and called Upper Lake Mary, this reservoir became an important aspect of Flagstaff's municipal water supply. Late in 1950, Upper Lake Mary dam was raised 3.6 m, increasing the reservoir depth to 11.7 m (Miller, 1954) and the total capacity to more than 5.1 billion gallons. Mean annual inflow to Upper Lake Mary has been estimated at 7,770 ac-ft per year. The mean annual seepage from the Upper Lake Mary reservoir is about 3,190 ac-ft per year (Harshberger and Associates 1977). Reports indicate that between 1904 and 1973 water has flowed through the canyon five times (Walnut Canyon National Monument 1976). Summer and early fall flows sufficient to cause flooding of the Walnut Canyon drainage have been rare events and have only been documented in 1972, 1987, and 1991.

Information on prehistoric vegetation communities is scarce. However, a paleobotanical study comparing past and present vegetation analyzed and radiocarbon dated eight packrat (*Neotoma*) middens collected in Walnut Canyon (Murdock, 1994). They ranged in age from 70 to 3,800 years BP. The plant remains of the middens were compositionally and proportionally similar to the modern vegetation, except for the oldest midden, in which conifer needles were far more common. The abundance of yucca, snakeweed, rabbitbrush, sage, and buckwheat has increased over time. Little to no yucca was found in the three oldest middens, which dated between 3,430 and 3,800 year BP. Murdock (1994) postulates that the Sinagua Indians may have introduced, cultivated, and harvested yucca between 860 and 790 years ago.

Geology

The exposed stratigraphic succession in Walnut Canyon consists of 225 m of Permian Coconino Sandstone overlain by 111 m of Permian Kaibab Formation. The Triassic Moenkopi Formation overlying these strata ranges in thickness from 0 to 120 m. The formation is present in isolated outcrops along the upthrown side of the Anderson Mesa Fault in Walnut Canyon (Peacock 1978) and on the Anderson Mesa adjacent to drainages leading into Walnut Canyon. Lava flow and cinder deposits are common and they cap Anderson Mesa and much of the Lake Mary watershed area. Within the Lake Mary Graben, the lava flows are approximately 38 m thick and are covered by Quaternary alluvium. Colluvial deposits are primarily found along the downthrown sides of faults, in canyons, and along margins of lava flows (Peacock 1978). Their thickness has been estimated to exceed 445 m (Harshberger and Associates 1977).

Structurally, the area is dominated by faults of hydrological significance. The major Anderson Mesa Fault forms the southwest boundary of Anderson Mesa and the northeast boundary of Lake Mary Graben. This fault strikes northwest along Upper and Lower Lake Mary before turning due north beyond the southern edge of Upper Lake Mary and the north-south trending part of Walnut Canyon (Peacock 1978). Approximately 1 mile south of Fisher Point the fault cuts across Anderson Mesa, and crosses and exits Walnut Canyon directly below Fisher Point. Numerous small faults, which may postdate the Anderson Mesa Fault, strike parallel to it along the downthrown side and delineate the Lake Mary Graben. The Lake Mary Fault, which is downthrown by 36 m (Peacock 1978), is the largest of these and lies adjacent to and southwest of the Anderson Mesa Fault.

Geohydrology

Walnut Canyon's morphology should be understood in terms of channel form and pattern. There is a close relationship between geologic structural features and drainage. The incised meanders in the northern part of Walnut Canyon indicate rejuvenation of the creek due to the uplift of the Colorado Plateau (Henkle 1976). These meanders were cut by the original stream bed configuration prior to uplift and remained in their orientation after uplift.

Tectonic disturbances have created numerous joints and fractures. Sinkholes and other expressions of solution activity are common along such fractures. Joint orientations in the Walnut Canyon area are probably inherited from the large north-south trending faults, some of which are presently active (Henkle 1976). Highly fractured rocks in the vicinity of the major faults transmit 10 to 50 times as much water as do the same rocks when unfractured. In the Kaibab Formation, the joints and faults have been widened by solution activity. Numerous such solution fea-

tures near the northwest end of Lower Lake Mary had to be dammed off because of the large quantities of lake water being lost (Akers 1962).

Since both Upper and Lower Lake Mary are located in a graben, much of the water stored is lost to seepage through fault fractures. This problem becomes aggravated in high-flow years as seepage loss rates increase with storage depth and lake surface area (Harshberger and Associates 1977; Blee 1988).

Surface stream flows in the study area fluctuate widely and most commonly occur during snowmelt and monsoon summer storms. However, the high infiltration capacity of the dry soil inhibits surface runoff, and little water is supplied to feeder streams during summer thunderstorms. Sudden melting of a high snow pack and ground saturation are required to yield significant runoff (Harshberger and Associates 1977). The mean inflow between 1941 to 1972 at Upper Lake Mary was estimated at 9.2 cfs, or 7,770 ac-ft per year (Blee 1975; Harshberger and Associates 1977) with a minimum of 0 and a maximum of 24.2 cfs, or 17,500 ac-ft per year (McGavock et al. 1986). Although no springs are known to occur in the canyon, several seeps have been observed. The seeps in the canyon walls may derive from locally recharged minor groundwater systems.

Hydrology

Walnut Canyon was an ephemeral drainage as indicated by historic photographs, literature, and anecdotal reports. Historically, flow occurred seasonally during snowmelt runoff in early spring and monsoonal thunderstorms in the summer. Ephemeral flow conditions are supported by the presence of 300-year-old ponderosa pine and Douglas fir trees on rocky terraces in the bottom of Walnut Canyon. Today these terraces are evidently a relatively stable component of the canyon bottom landscape in the Walnut Canyon drainage.

Inflow, lake level, and spill data are lacking or sporadic. Long-term data for lake level, gross lake volume, and surface water diversion exist only for Upper Lake Mary. Reservoir construction and storage has altered the frequency of flows in Walnut Canyon and has probably altered the magnitude and duration of the flood peaks. A simple mass balance model indicates that since 1950, flow events through Walnut Canyon have been reduced from almost an annual event to 1 in every 9 years, due to the construction of Upper and Lower Lake Mary. We have calculated that since records were kept (1950–1993) total flow volume into Walnut Canyon has been reduced from an estimated 290,000 ac-ft to about 25,000 ac-ft due to the presence of a single reservoir, Upper Lake Mary. It has probably been reduced another order of magnitude by the construction of both reservoirs.

When Upper Lake Mary overflows, a high-flow event may occur in Walnut Canyon provided that the duration and magnitude of the overflow event are sufficient to fill up Lower Lake Mary, cause it to overflow, and in turn create a flow of sufficient magnitude to overcome water losses through fault fractures and bank storage. Five major overflow events (longer than 5 weeks duration) from Lower Lake Mary have been recorded, according to Flagstaff Water Treatment Plant records. These overflow events occurred in 1973, 1979, 1980, 1983, and 1993.

The timing of spring runoff and/or high flows has also been altered by reservoirs and surface diversion to the city of Flagstaff. Anecdotal and photographic evidence indicates that summer flows from the upper reaches of the stream have been completely eliminated, and only local runoff events, generated by the tributaries to Walnut Canyon, passed through Walnut Canyon National Monument. Such flows have been documented in 1972, 1987, and 1991.

Dendrochronology

Only two species, ponderosa pine and Arizona walnut, had individuals that were of sufficient age to yield usable tree-ring data. The mean tree-ring index of ponderosa pine cores collected from two habitats—the rims and south facing slopes, and canyon bottom—were calculated over three critical time periods: prior to 1904, before any dams were in place; 1905 to 1941, after the construction of Lower Lake Mary dam and before the construction of Upper Lake Mary dam; and 1942 to 1992, after both dams were built. Rim/slope tree-ring index means were employed as a reference to compare similar values for canyon-bottom ponderosa pines.

The mean tree-ring index of ponderosa pine trees on the canyon rim and upper slopes has changed when calculated over the three defined time periods (Figure 2). In comparison, tree-ring indices of ponderosa pine trees in the bottom of

the canyon appear to have changed little over time in response to construction of the Lake Mary dams and subsequent dewatering of the drainage. The period 1942–1985, especially during the 1950s, was one of drought in the Southwest (Neilsen 1986), and growth rates of the more sensitive rim-slope trees declined in response. In contrast, the slower growing canyon-bottom ponderosa pines situated in complacent sites were less responsive to changes in climate.

No evidence of dam-induced dewatering was observed on the growth of either ponderosa pine or Arizona walnut growing in the canyon. There was no substantive change in the mean tree-ring index of ponderosa pine in and directly adjacent to the canyon bottom during the period prior to construction of Lower Lake Mary dam and after both dams were in place. In fact growth rates, as represented by the mean ring index, actually rose somewhat between 1942 and 1992 (0.98 to 1.05). The increase in the mean tree-ring index over this time period was even more pronounced for Arizona walnut (0.87–1.06) (Figure 3d). In summary there is no compelling evidence that the construction of the two dams in the Walnut Canyon drainage has had any effect on the growth rates of either Arizona walnut or ponderosa pine trees growing in the bottom of Walnut Canyon as reflected in changes in the mean tree-ring index.

Vegetation Studies

The modern vegetation in Walnut Canyon has been described by six studies (Arnberger 1947; Spangle 1953; Joyce 1974, 1976; Phillips 1990; Jenkins et al. 1991). Five plant communities have been described in the canyon (Jenkins et al. 1991): (a) A ponderosa pine forest on the canyon's rims covers the largest area, followed by (b) a pinyon–juniper conifer woodland. Three associates are present within the canyon: (c) the south-facing slopes support a yucca–pinyon pine–blue grama woodland, (d) the north-facing slopes are covered by a Douglas fir–Gambel oak–muttongrass forest, and (e) a deciduous, riparian woodland is present on the canyon bottom. The latter is divided into two subassociations: boxelder–wormwood–Arizona rose–New Mexico locust, and boxelder–narrowleaf cottonwood.

A canyon-bottom inventory of Walnut Canyon (Phillips 1990) lists 155 species in 127 genera and 51 families based upon collections made in 1989 and existing herbarium records. The bottom of Walnut Canyon has the greatest diversity and contains over half the species when compared to upland plant communities. Thirteen species, or 8 percent of the flora, are exotic plants, generally introduced from Europe or Eurasia. A partial list of the most important canyon-bottom species—those encountered in the authors' sampling plots—is presented in Table 1. A complete list can be found in Phillips (1990).

Historic photographs suggest that there has been an increase in vegetation since the 1940s when Upper Lake Mary dam was built (Brian 1992). There are no reliable data available that document the status of the canyon-bottom vegetation prior to this time period. In fact, the first vegetation survey of the Walnut Canyon bottom was done by Joyce (1974, 1976) and he did not sample within the National Monument. However, we suspect that before the Lake Mary dams were in place, constant and short seasonal flows were sufficient to maintain a stream bed largely clear of vegetation. A narrow band of riparian vegetation would have occurred on either side of the largely barren, boulder-strewn stream bottom. This is the present condition of nearby undammed drainages.

Based on their size–frequency distribution (Figure 4), three of the four important tree species occupying the bottom of Walnut Canyon, Arizona walnut, Rocky Mountain juniper and Gamble oak, are perpetually invading the stream channel. The often long period of time between high-flow events allows seedlings of these species to germinate and become established. Large numbers of these trees are then eliminated by the next major flow and if a few survive, as for example appears to be the case for Rocky Mountain juniper, they are eliminated by a subsequent flow. By observation, larger, more mature trees of these species tend to occur on the periphery of the channel or on raised gravel bars protected from all but the largest flows. Boxelder trees in Walnut Canyon are slow growing and appear to be reproducing mostly from resprouts.

Comparison Canyons

Five drainage systems were investigated for comparison studies: West Fork of Oak Creek Canyon, Fry Canyon, Padre Canyon, Mormon Canyon, and Sycamore and Volunteer canyons. The most noticeable difference between Walnut Canyon and nearby, adjacent undammed drain-

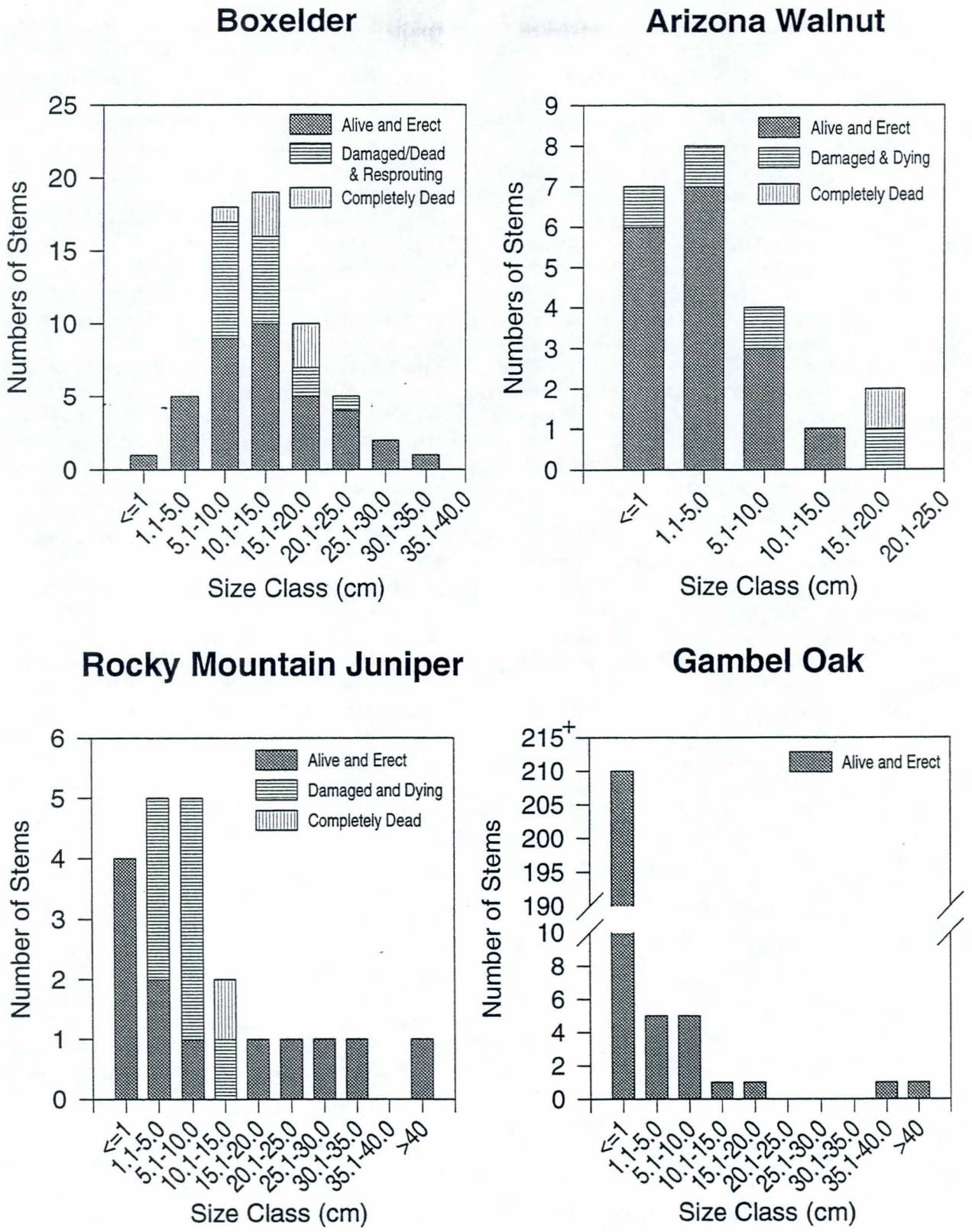


Figure 4. Size class distributions for four tree species growing within the stream channel in the bottom of Walnut Canyon within Walnut Canyon National Monument.

ages is in the distribution of vegetation relative to the stream channel and the substrate composition of the channel itself. Comparison drainages always had open channels, or nearly so, and were easily navigable on foot, as compared with long reaches of the Walnut Canyon drainage which was often choked with shrubby invader species. New Mexico locust and other thicket formers (red osier dogwood, Arizona rose, etc.) should be largely restricted to adjacent banks and slopes, away from the channel proper, as they are in the comparison drainages. Comparison drainages had few silt deposits. Instead their channels consisted largely of cobble and boulder-sized debris with very few extensive silt accumulations. Silt accumulations are very common throughout the Walnut Canyon drainage below Lower Lake Mary and are the result of sediment deposition from erratic dam overflows. These sediment deposits form a suitable substrate for invasion of exotic annual and short-lived perennial plants.

A comparison of nearby undammed tributaries shows that Walnut Canyon would be able to support a more typical riparian ecosystem if it had consistent flows every year. Furthermore, the channel should be relatively free of vegetation. Some noticeable differences between the vegetation composition of the canyon bottom and stream channel in West Fork of Oak Creek Canyon and Walnut Canyon are:

- A higher relative cover of willows (*Salix* spp.), Douglas fir, and red osier dogwood occurs in West Fork of Oak Creek Canyon; willows are present in Walnut Canyon only at seeps.
- A higher cover of boxelder, narrowleaf cottonwood, Arizona walnut, snowberry, dragonsage, New Mexico locust, perennial grasses and exotic invader plant species occurs in Walnut Canyon.
- Big-tooth maple is generally absent from the Walnut Canyon drainage bottom.

Similar differences were found when comparing the bottom vegetation of Walnut Canyon with Fry Canyon. Because of the irregular flooding and drought within Walnut Canyon, we surmise that willows can no longer compete with more aggressive shrubby increaser species along the stream channel.

Species such as Arizona walnut, narrowleaf cottonwood, and snowberry, which are relatively important in the Walnut Canyon bottom vegetation, are not prevalent associates in Fry Canyon or the upper reaches of West Fork of Oak Creek Canyon, although both species occur within these drainages. Snowberry is a thicket-forming increaser species, so its abundance in the disturbed Walnut Canyon stream bottom and absence in Fry and West Fork of Oak Creek canyons is explainable.

The most noticeable difference in Padre Canyon is the greater preponderance of willow species and the more open nature of the stream channel. As in Fry and West Fork of Oak Creek canyons, and unlike Walnut Canyon, the stream bottom is never choked by vegetation. In Mormon Canyon the drainage is ephemeral, dependent upon snowmelt and local thunderstorms. Gambel oak and ponderosa pine comprise the tree overstory together with an occasional one-seed juniper. Utah serviceberry contributes to the shrub layer. The ground layer is dominated by Arizona rose. No willows or other obligate riparian species were observed at Mormon Canyon.

True riparian, hydrophilic species, such as broad-leaved cat-tail and yellow pond lily, are absent from Walnut Canyon but have been described from Sycamore Canyon (Schilling 1980). Volunteer Canyon, a tributary to upper Sycamore Canyon, shares many of the dominant species with Walnut Canyon (Douglas fir, ponderosa pine, red osier dogwood, boxelder, Rocky Mountain juniper, willow species, and meadow rue) but does not have the aquatic species that were found in Sycamore Canyon.

Effect of High Flows on Vegetation

The impact of water flow in Walnut Canyon varies depending on whether the flow occurs during the dormant winter season or the active growing spring and summer seasons. The effect of flooding on bottomland species is greatly influenced by five critical factors: time of year, flood frequency, flood duration, water depth, and siltation (Teskey and Hinckley 1977; Malanson 1993). Understory vegetation is strongly influenced by flooding, with an increase in herbaceous species diversity as flood frequency decreases (Bell 1974).

The vegetation in the dry bottom of Walnut Canyon consists of a great many shrub, subshrub, and woody liana species. Many of these, such as New Mexico locust, Arizona rose, and red osier dogwood, are active resprouters, while an associate, desert olive, is a thicket former (Lit-

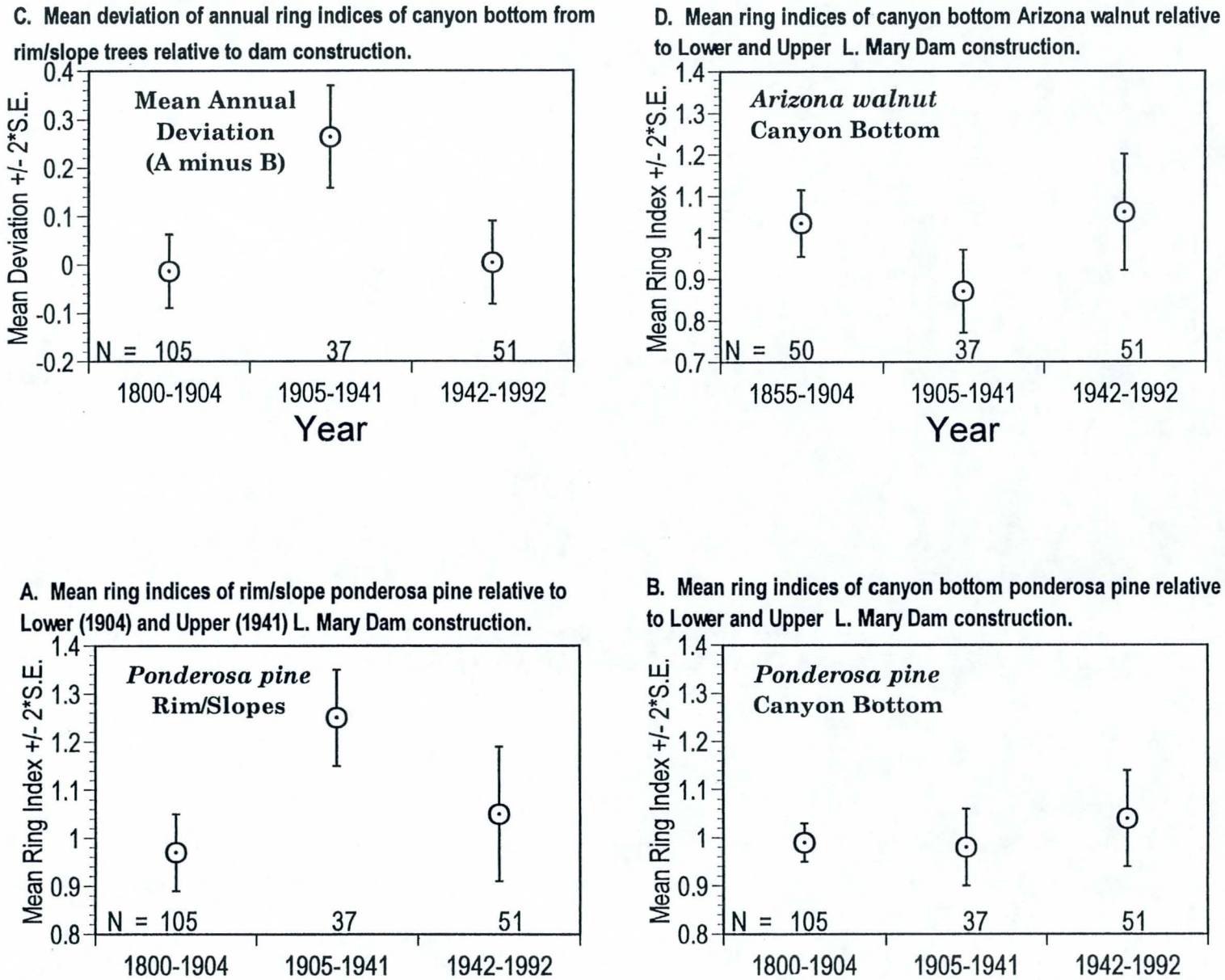


Figure 3. Changes in the mean tree index for ponderosa pine (rim/slope and canyon bottom sites) and Arizona walnut (canyon bottom) over discrete time periods associated with the construction of Upper and Lower Lake Mary Dams.

Table 1. Alphabetical List, by Common Name, of Common Plant Species Found in Walnut Canyon National Monument.

Common Name	Native or Exotic	Scientific Name
Apache plume	Native	*Fallugia paradoxa*
Arizona grape	Native	*Vitus arizonica*
Arizona rose	Native	*Rosa arizonica*
Arizona walnut	Native	*Juglans major*
Aspen	Native	*Populus tremuloides*
Bee balm	Native	*Monarda menthaefolia*
Blue grama	Native	*Bouteloua gracilis*
Bonpland willow	Native	*Salix bonplandiana*
Boxelder	Native	*Acer negundo*
Buckwheat	Native	*Erigonum* spp.
Canada wild rye	Native	*Elymus canadensis*
Clematis	Native	*Clematis ligusticifolia*
Cliffrose	Native	*Cowania stansburiana*
Desert olive	Native	*Forestiera pubescens*
Dogbane	Native	*Apocynum cannabinum*
Douglas fir	Native	*Pseudotsuga menziesii*
Downy chess	Exotic	*Bromus tectorum*
Dragon sage	Native	*Artemisia dracunculus*
False solomon seal	Native	*Smilicina racemosa*
Fernbush	Native	*Chamaebatiaria millefolium*
Four-wing saltbush	Native	*Atriplex canescens*
Fremont barberry	Native	*Berberis fremontii*
Gambel oak	Native	*Quercus gambelii*
Goldenrod	Native	*Solidago sparsiflora*
Hop	Native	*Humulus americana*
Little bluestem	Native	*Schizachyrium scoparium*
Magellans phacelia	Native	*Phacelia magellanica*
Meadow rue	Native	*Thalictrum fendleri*
Milk vetch	Native	*Astragalus tephrodes*
Missouri iris	Native	*Iris missouriensis*
Motherwort	Exotic	*Leonurus cardiaca*
Mutton grass	Native	*Poa fendleriana*
Narrowleaf cottonwood	Native	*Populus angustifolia*
Narrowleaf hoptree	Native	*Ptelea angustifolia*
New Mexico locust	Native	*Robinia neomexicana*
One-seed juniper	Native	*Juniperus monosperma*
Poison ivy	Native	*Toxicodendron radicans*
Ponderosa pine	Native	*Pinus ponderosa*
Rabbitbrush	Native	*Chrysothamnus* sp.
Red osier dogwood	Native	*Cornus stolonifera*
Richardsons brome	Native	*Bromus richardsonii*
Rocky Mountain juniper	Native	*Juniperus scopulorum*
Russian thistle	Exotic	*Salsoa iberica*
Sage	Native	*Artemisia* sp.
Scouler's willow	Native	*Salix scouleriana*
Sedge	Native	*Carex occidentalis*
Service berry	Native	*Amelanchier utahensis*
Sideoats grama	Native	*Bouteloua curtipendula*
Snakeweed	Native	*Gutierrezia* sp.
Snowberry	Native	*Symphoricarpos parishii*
Squaw bush	Native	*Rhus trilobata*
Thicket creeper	Native	*Parthenocissus vitacea*
Wax currant	Native	*Ribes aureum*
Wheatgrass	Native	*Agropyron smithii*
Willows	Native	*Salix* spp.
Wormwood	Native	*Artemisia ludoviciana*
Yucca	Native	*Yucca* sp.

tle 1976). The dewatering of the Walnut Canyon drainage appears to have encouraged invasion of these species into the abandoned stream channel. Judging from hydrological data documenting Lower Lake Mary overflows, flow events have not occurred with sufficient intensity or frequency to eliminate these invaders. The observed 1993 Lake Mary overflow appears to have had little impact on the canyon-bottom vegetation. Examination of previous vegetation data collected by Joyce (1974) and Philips (1990) as well as our own suggests that the vegetation cover composition of most species of trees and shrubs occupying the bottom of Walnut Canyon has remained more or less constant over time, in spite of the several major dam overflows that occurred in 1973, 1979, 1980, 1983 and 1993.

Conclusion

The impacts of geologic fracturing and faulting on channel transmission and storage of water in Walnut Canyon cannot be quantified, but are suspected to be large. Water losses from the reservoirs are well documented. Discharge of natural seeps in Walnut Canyon may be enhanced by upstream reservoir storage and may promote the maintenance of obligate riparian species.

Comparative studies of nearby tributaries such as Fry Canyon and West Fork of Oak Creek Canyon indicate that these undammed channels receive sufficient seasonal runoff to allow predictable semiannual flows that are largely absent from Walnut Canyon. The annual runoff events in these drainages seem to maintain channels that are more open and that contain more pools and less silty deposits. It is believed that these conditions promote the maintenance of naturally occurring vegetation including obligate riparian plant species.

The plants of the dry Walnut Canyon bottom constitute "transitional" vegetation that is maintained as a result of the irregular and unpredictable high-flow events resulting from winter/spring overflows of Upper and Lower Lake Mary. Such events are relatively rare in the summer and early fall when canyon-bottom flows may result from local runoff due to localized monsoonal storms or tropical disturbances.

The low rate of disturbance is enough to remove non-flood and non-disturbance-adapted vegetation from the bottom of Walnut Canyon. However, the canyon remains essentially dry most of the time. Major winter and spring flows may be absent for as long as a decade. The open water that exists is largely the result of snowmelt and monsoonal precipitation, which temporarily fills pools. However, throughout most of the canyon bottom, such long periods of imposed drought effectively eliminate obligate riparian species, such as some species of willows, which are now restricted in the canyon bottom to the vicinity of seeps.

Even a relatively severe, high-flow event such as occurred in the winter and spring of 1993 does not totally remove all the vegetation that has invaded the canyon bottom. Disturbance-adapted species such as New Mexico locust, red osier dogwood, boxelder, and Arizona rose were observed to be resprouting vigorously. Boxelder appears to be reproducing primarily through vegetative propagation and seedling establishment is sparse. The vegetation is generally dormant at the time of year when these high flows are most likely to occur and the extremely cold runoff would tend to reduce a plant's metabolic activity, thereby enabling roots to survive a relatively long period of inundation.

Rocky Mountain juniper and other species with a low flood tolerance will be periodically removed or damaged by consecutive high flows. Narrowleaf cottonwoods have short-lived seeds that require full sunlight and a saturated soil for an extended period of time for germination (Vines 1960; Lanner 1984). Because of the now irregular and rare high flows, these may be in the process of removal from the system.

The slight increase from 1989 to 1993 of New Mexico locust, especially after the runoff event of 1993, attests to the hardiness of this leguminous shrub or small tree species. The fast growth rate, accompanied by the tendency of this species to root sprout, is an adaptation to disturbances such as irregular flooding (Johnson 1993), and increases the likelihood that New Mexico locust will continue to form thickets and colonize the canyon bottom.

Bare sandy areas are invaded by weedy herbs and forbs, many of them introduced species. Plant species that are not adaptable to disturbance-creating, high-flow events by resprouting, rapid germination and establishment, or that are unable to survive a long drought through tapping ground water sources, as well as annual life forms will eventually be eliminated from the bottom of the drainage system. Past and present plant collections have shown that hydrophilic

plants such as marsh smartweed no longer grow in Walnut Canyon.

The low cover of Bonpland willow and arroyo willow and the absence of hydrophytes throughout the corridor of Walnut Canyon is a direct result of the lack of flowing water. Only those obligate riparian species that can inhabit areas below or at seeps or that are dependent upon locations where slope runoff is channeled can survive. In summary, the vegetation seen occupying the various substrates in the bottom of Walnut Canyon is a cyclical, transitional vegetation surviving and maintaining itself because of the perpetual disturbance by erratic dam overflows.

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SUSTAINABILITY OF FISHES IN DESERT RIVER: PRELIMINARY OBSERVATIONS ON THE ROLES OF STREAMFLOW AND INTRODUCED FISHES

Jerome A. Stefferud[1] *and John N. Rinne*[2]

Patterns of stream discharge and the presence of introduced fishes appear to interact to influence community dynamics and sustainability of native fishes in streams of the arid Southwest (Minckley and Meffe 1987). A long history of water development and diversion coupled with numerous introductions of non-native and exotic fishes has resulted in few streams in Arizona retaining predominantly native fish communities (Miller 1960; Minckley and Douglas 1991). Although native fishes yet occur in most drainages in the state, the number of species occurring in any stream typically is reduced from historic levels. Where native fishes are present, the community dynamics apparently are altered by non-native or exotic species (Minckley and Deacon 1968; Rinne and Minckley 1991; Deacon and Minckley 1991).

Minckley and Meffe (1987) contrasted the differences in flood hydrology between southwestern arid lands and those of lowland mesic regions of the central and eastern U.S. They found that most of the annual water yield in southwestern systems was produced during high discharges for brief periods of time, whereas low discharges produced a far greater proportion of total yield from mesic systems. The differential effects of these patterns on fishes have had a significant impact on the native-introduced mix of species now present in the Southwest. Minckley and Meffe (1987) concluded that native southwestern fishes were better adapted to withstanding the effects of large floods than non-native fishes. During significant floods, non-native species were either displaced or destroyed, whereas native species maintained position in or adjacent to channel habitats, persisted in microrefuges, or rapidly recolonized if displaced.

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The upper Verde River, central Arizona, from its source at Sullivan Lake to the mouth of Sycamore Creek 60 km downstream, is rare among Arizona rivers in that it still retains its native fish fauna. The six native fishes historically recorded there still occur abundantly, and at least seven non-native species are established, with many others reported. The native fish fauna includes longfin dace (*Agosia chrysogaster*), speckled dace (*Rhinichthys osculus*), roundtail chub (*Gila robusta*), spikedace (*Meda fulgida*), desert sucker (*Pantosteus clarki*), and Sonora sucker (*Catostomus insignis*) (Minckley 1973; Minckley 1985). The skeletal remains of razorback sucker (*Xyrauchen texanus*) and Colorado squawfish (*Ptychocheilus lucius*) were found at an archaeological site near Perkinsville (dated ca. 1300–1400 A.D.) (Minckley and Alger 1968), and both species have recently been stocked in this reach (Hendrickson 1993). Non-native species commonly found there include red shiner (*Cyprinella lutrensis*), fathead minnow (*Pimephales promelas*), common carp (*Cyprinus carpio*), green sunfish (*Lepomis macrochirus*), smallmouth bass (*Micropterus dolomieu*), channel catfish (*Ictalurus punctatus*), and yellow bullhead (*Ameiurus natalis*) (Minckley 1973; Minckley 1985; Hendrickson 1993). Reasons for retention of the native fish fauna are unclear, but may be related to the canyon-bound nature of the river and its relatively unregulated flows that can scour the channel and evidently affect the dynamics of the fish community (Minckley 1985; Sullivan and Richardson 1992).

Many studies of the fishes and aquatic ecology of the upper Verde River are available in the published literature and in scattered agency reports, but none have investigated the long-term interrelationships of the fish community with disturbance events. In order to gain an understanding of how the fish community in the upper Verde River responds to disturbance events (drought, flooding, invasions by non-native species), we initiated a study in 1994 to

observe changes in species abundance and occurrence through time and space, fish densities and biomass relative to aquatic habitat types, and mobility of fishes. Long-term and consistent monitoring of the fishery at several sites along the length of the upper Verde River will be accomplished annually during a 10-year period. The objective of the study is to determine the roles and relative influence of physical (hydrology and geomorphology) and biological (introduced fish population dynamics, fish mobility, native versus non-native interactions, reintroduction of native species) factors in the sustainability of native fishes in time (10 years) and through space (7 sites over 60 km). Our purpose herein is to report preliminary results from our first year of sampling.

Methods

The general location of study sites was chosen based primarily on reasonable access and linear disposition over the 60-km reach. Specific locations were based on channel morphology and types of habitat present. Access to this reach of the Verde River, in particular the 20 km east of Perkinsville, is limited by the topography of the watershed and lack of roads. There is only one bridge across the river (at Perkinsville), and access to the valley bottom at other sites is limited to four-wheel-drive vehicles or foot trail. The canyon bottom is administratively restricted by the Forest Service to non-motorized travel, although the Topeka and Santa Fe Railroad parallels the river downstream from Perkinsville. Seven sites were chosen starting at a point 7.6 km east of Sullivan Lake and ending at the confluence of Sycamore Creek (Figure 1). Sites were linearly separated by distances of 3.3 to 16.2 km.

We proposed to sample 250 to 300 m of stream at each site, but exact length was dependent on habitat complexity at the site. In general, a riffle-pool sequence was selected, and then the study site was expanded to include other significant habitat types present. Habitat types were defined by combinations of gradient, subjective descriptions of channel shape and turbulence, and relative position in the stream channel (Bisson et al. 1982; Aadland 1993).

Habitat types were sampled for fish in sequence, progressing from downstream to upstream. The methods used were appropriate to the habitats sampled, including backpack DC electrofishing units, seines (3.2 mm mesh), dipnets, and trammel nets. All fish captured within a habitat type were identified to species and enumerated. Measurements of total length (TL) were obtained from a subsample of each species. Fish were then returned alive to the stream.

Each habitat type was numbered in sequence and named (lateral scour pool, mid-channel pool, backwater pool, glide, run, low and high gradient riffles, and edgewater). Length, width, depth, velocity, and substrate for each habitat type were recorded. For future reference, a map of the entire site was sketched and location of habitat types noted. Water chemistry was recorded, including dissolved oxygen, conductivity, total dissolved solids, pH, turbidity, and temperature. Total length of the site was measured, and notes on the shape of the valley bottom, width of the active channel, and land uses were made.

Results

Sixty-seven habitat types at seven sites were sampled for fish and measured for habitat characteristics during April and May 1994. In October 1994, 12 selected habitat types at Burnt Ranch and Sycamore sites were resampled for fish. Thirteen species (8,705 individuals) were captured during the spring sampling (Table 1), and 11 species (1,148 individuals) in the autumn (Table 2). The total number of individuals captured per site ranged from 451 to 2,837. The number of native species comprised 82 percent of the spring sample, but only 54 percent of the autumn sample. The non-native red shiner accounted for 17 percent of the total spring fish catch, and 39 percent of the autumn. Seining accounted for 57 percent, and electrofishing 43 percent, of the total catch in spring. Only seining was used during autumn sampling.

The relative abundance of native and non-native species varied longitudinally and seasonally. Longfin dace was common at the upper two sites but uncommon or absent at the lower sites, whereas the opposite pattern applied to speckled dace. Roundtail chub was relatively common at the lower four sites in the spring, but was found in greater abundance at the upper site in the autumn. The threatened spikedace was abundant at the upper site, extremely rare at the next three sites downstream, and relatively common at the lower three sites during spring sampling. However, it was absent from

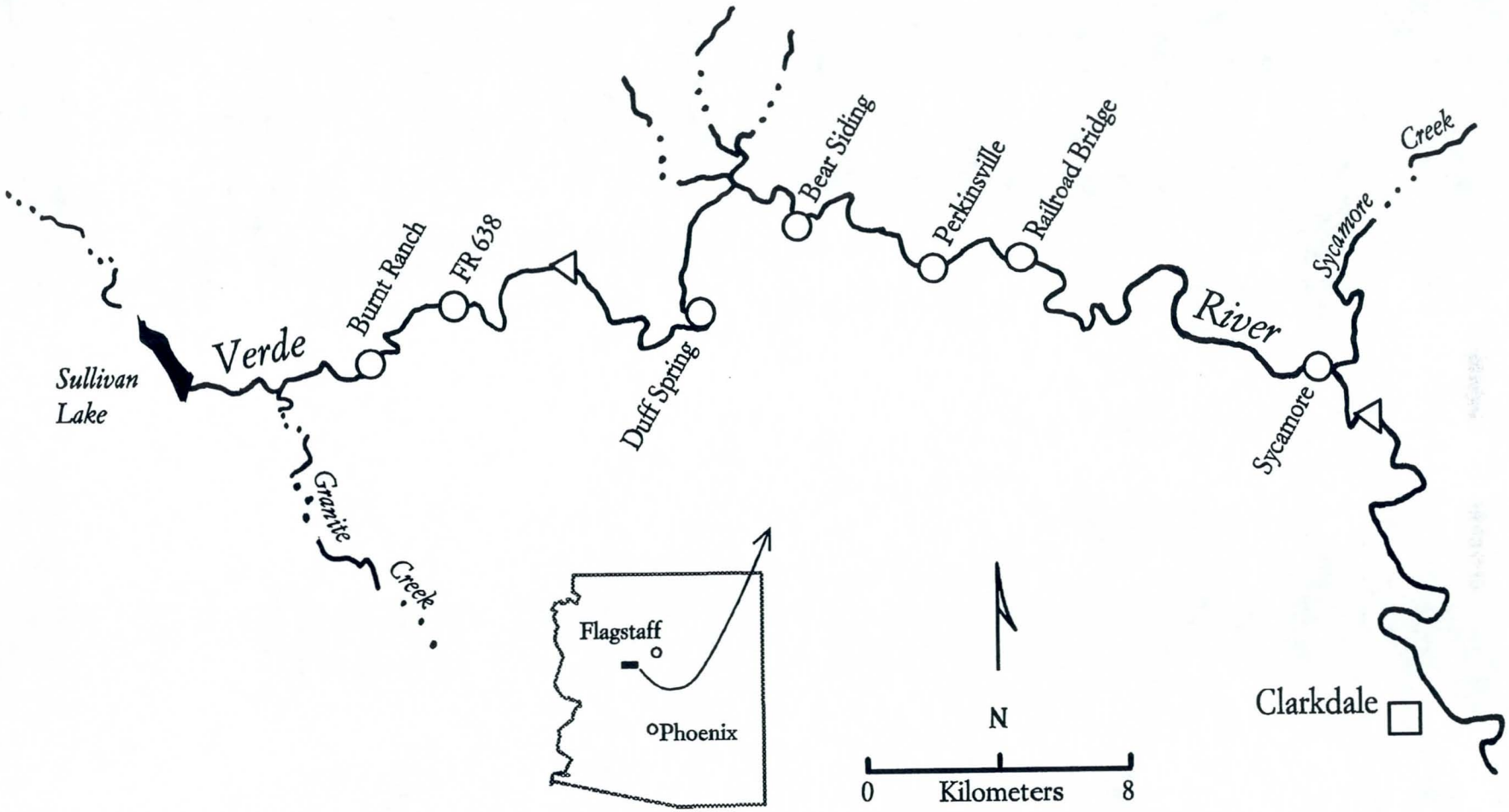

Figure 1. Map of study area with open circles indicating sample sites. Triangles indicate locations of USGS Paulden and Clarkdale gages.

Table 1. Summary of fish collections from Upper Verde River, April–May 1994 (number and relative abundance as percentage).

Species	Burnt Ranch	FR638	Duff Spring	Bear Siding	Perkins-ville	Railroad Bridge	Sycamore	Total
Longfin dace	1072 (53.5)	227 (32.7)	--	--	19 (0.7)	--	1 (0.1)	1319 (15.2)
Desert sucker	339 (16.9)	230 (33.1)	192 (32.5)	382 (31.1)	885 (31.2)	237 (52.5)	379 (42.2)	2644 (30.4)
Sonora sucker	278 (13.9)	154 (22.2)	329 (55.8)	357 (29.1)	442 (15.6)	27 (6.0)	223 (24.8)	1810 (20.8)
Roundtail chub	15 (0.7)	18 (2.6)	28 (4.7)	249 (20.3)	244 (8.6)	57 (12.6)	165 (18.4)	776 (8.9)
Spike dace	257 (12.8)	2 (0.3)	1 (0.2)	1 (0.1)	37 (1.3)	38 (8.4)	92 (10.2)	428 (4.9)
Speckled dace	--	--	--	3 (0.2)	61 (2.2)	88 (19.5)	19 (2.1)	171 (2.0)
Yellow bullhead	2 (0.1)	3 (0.4)	1 (0.2)	1 (0.1)	12 (0.4)	2 (0.4)	10 (1.1)	31 (0.4)
Common carp	1 (T)	--	3 (0.5)	4 (0.3)	11 (0.4)	--	4 (0.4)	23 (0.3)
Red shiner	39 (1.9)	61 (8.8)	32 (5.4)	227 (18.5)	1109 (39.1)	2 (0.4)	3 (0.3)	1473 (16.9)
Channel catfish	--	--	--	1 (0.1)	4 (0.1)	--	--	5 (0.1)
Green sunfish	--	--	--	--	4 (0.1)	--	--	5(T)
Smallmouth bass	2 (0.1)	--	4 (0.7)	3 (0.1)	2 (0.1)	--	3 (0.3)	14 (0.2)
Fathead minnow	--	--	--	--	7 (0.2)	--	--	7 (0.1)
N	2005	695	590	1228	2837	451	899	8705

Table 2. Summary of fish collections from Upper Verde River, October 1994 (number and relative abundance as percentage, in parentheses).

Species	Burnt Ranch	Sycamore	Total
Longfin dace	94 (15.5)	–	94 (8.2)
Desert sucker	31 (5.1)	93 (17.2)	124 (10.8)
Sonora sucker	214 (35.3)	25 (4.6)	239 (20.8)
Roundtail chub	50 (8.2)	17 (3.1)	67 (5.8)
Spike dace	93 (15.3)	–	93 (8.1)
Speckled dace	–	1 (0.2)	1 (T)
Yellow bullhead	1 (0.2)	6 (1.1)	7 (0.6)
Common carp	67 (8.2)	1 (0.2)	68 (5.9)
Red shiner	50 (8.2)	395 (73.0)	448 (38.8)
Channel catfish	–	–	–
Green sunfish	–	–	–
Smallmouth bass	2 (0.3)	3 (0.6)	5 (0.4)
Fathead minnow	5 (0.8)	–	5 (0.4)
N	607	541	1148

the lower site during autumn sampling, whereas red shiner comprised the majority of the autumn sample at that site.

Roundtail chubs in October (n = 67; 98 to 420 mm TL; x = 110.2 mm TL) were significantly greater in length ($t = -12.6372$; $P < 0.05$) than those in spring (n = 456; 38 to 433 mm TL; x = 99.5 mm TL).

Length of sites sampled during spring ranged from 124 to 361 m, and the amount of surface area per site ranged from 1,185 to 2,279 m^2. In autumn the length of sites sampled was 158 to 270 m and surface area was 993 to 439 m^2. Among these sites, 67 habitat types were sampled in spring and 12 in autumn. Habitat types were predominantly riffles (36%), followed by runs (25%), glides (16%), lateral scour pools (15%), and backwater pools (4%). Abundance of fish was highest in backwater pools, and was mostly attributable to the presence of Sonora sucker and roundtail chub (Figure 2). Longfin dace and spikedace were only in riffle, run, or glide habitats, and speckled dace were primarily

in riffle habitats. Desert sucker were in all habitat types. Red shiners were in glide and run habitats.

Discussion

Baseflow in the upper Verde River is primarily derived from springs and subsurface input; there are few tributaries and most are intermittent. Stream discharge is relatively constant at a site and is significantly affected only during extreme runoff or droughts. At the Paulden gage annual mean discharge is 1.3 m^3/sec, with monthly means between 0.7 and 3.7 m^3/sec. Downstream at the Clarkdale gage annual mean discharge is 5.7 m^3/sec, and monthly means are between 2.2 and 14.7 m^3/sec (Smith et al. 1993).

As is typical of desert streams and rivers, flooding can be significant in the Verde River (Figure 3). During January–February 1993 at Paulden, a peak discharge of 657 m^3/sec was preceded by three floods that each exceeded 200 m^3/sec. A peak discharge of 1,500 m^3/sec at Clarkdale was also preceded by three floods, each exceeding 500 m^3/sec (Smith et al. 1993). The peak floods, estimated to have recurrence intervals of 50 to 60 years (U.S. Geological Survey 1992), "reset" the biological and geomorphological reference points baseline. That is, riparian vegetational succession was set back to a base level, channel morphology was modified, and stream bed materials were sorted and rearranged.

The floods probably benefited native fish by breaking up embedded bottom materials (Mueller 1984) and reducing the populations of non-native fishes (Minckley and Meffe 1987). Observations in summer 1993 indicated that roundtail chub, spikedace, and desert and Sonora sucker had high reproductive success that year. Our collections in 1994 showed a strong 1 year age class for those species. With the exception of red shiner, non-native fishes comprised less than 1 percent of the total catch, although they may be under-represented because our sampling methods do not adequately sample habitats (e.g., deep pools) where non-native predators are more likely to occur. Red shiners were present at all sites but were common or abundant only at the middle two sites, and were a minor component of the fish fauna at the others.

During February 1995, the Verde River experienced floods in the 8 to 10 year recurrence intervals. Casual observations at sampling sites on March 29–30 under extremely clear water suggested a drastic reduction in all fishes in the sampling reaches. Although our study is focused on the effects of disturbances to the system, we did not anticipate record floods occurring during the initial stages of the project. Results of sampling this year and during out-years will make an interesting comparison with our present knowledge of the system.

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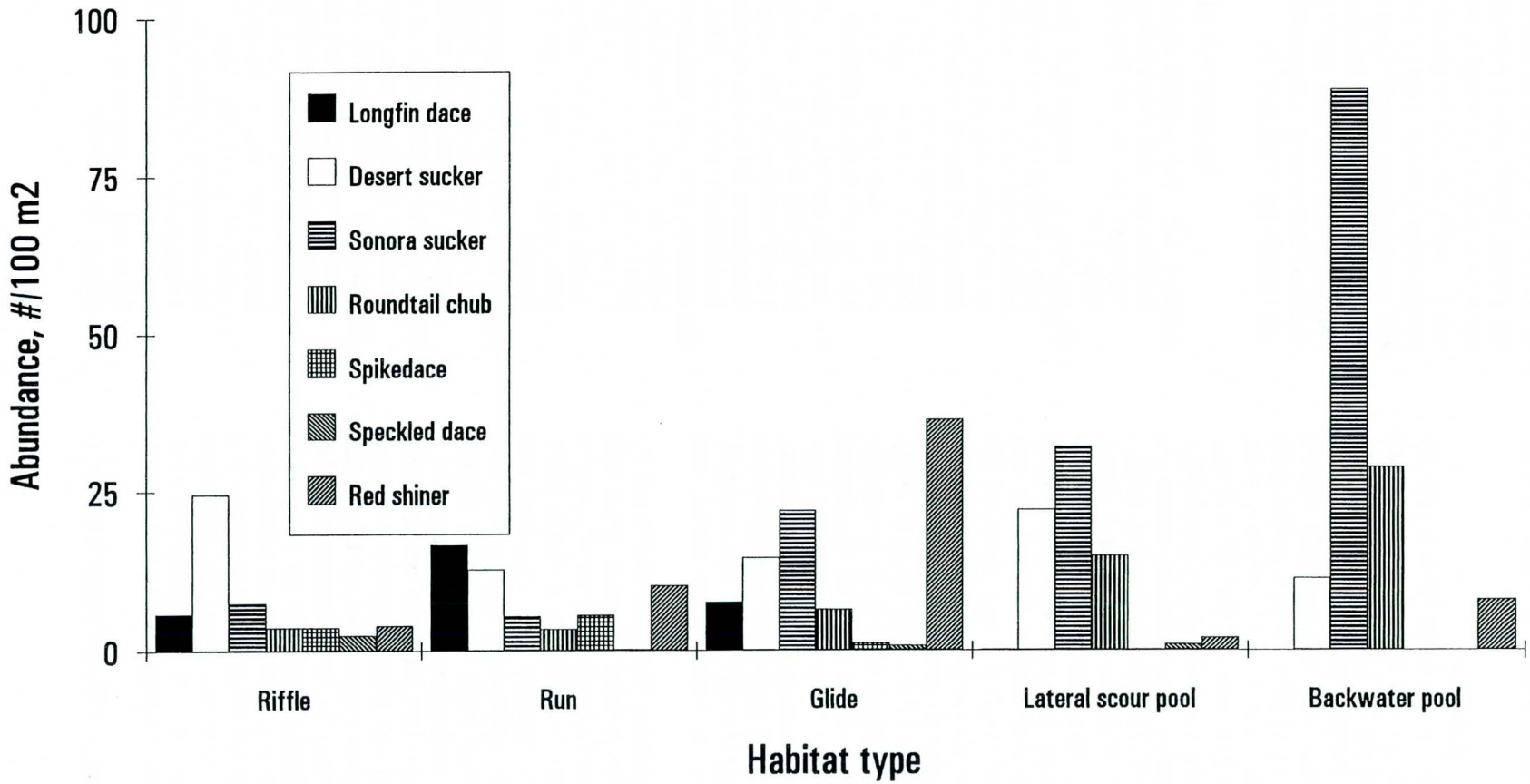

Figure 2. Abundance (number of individuals per 100 m^2 of surface area sampled) of selected fishes in major habitat types in Verde River captured during sampling in spring and autumn, 1993.

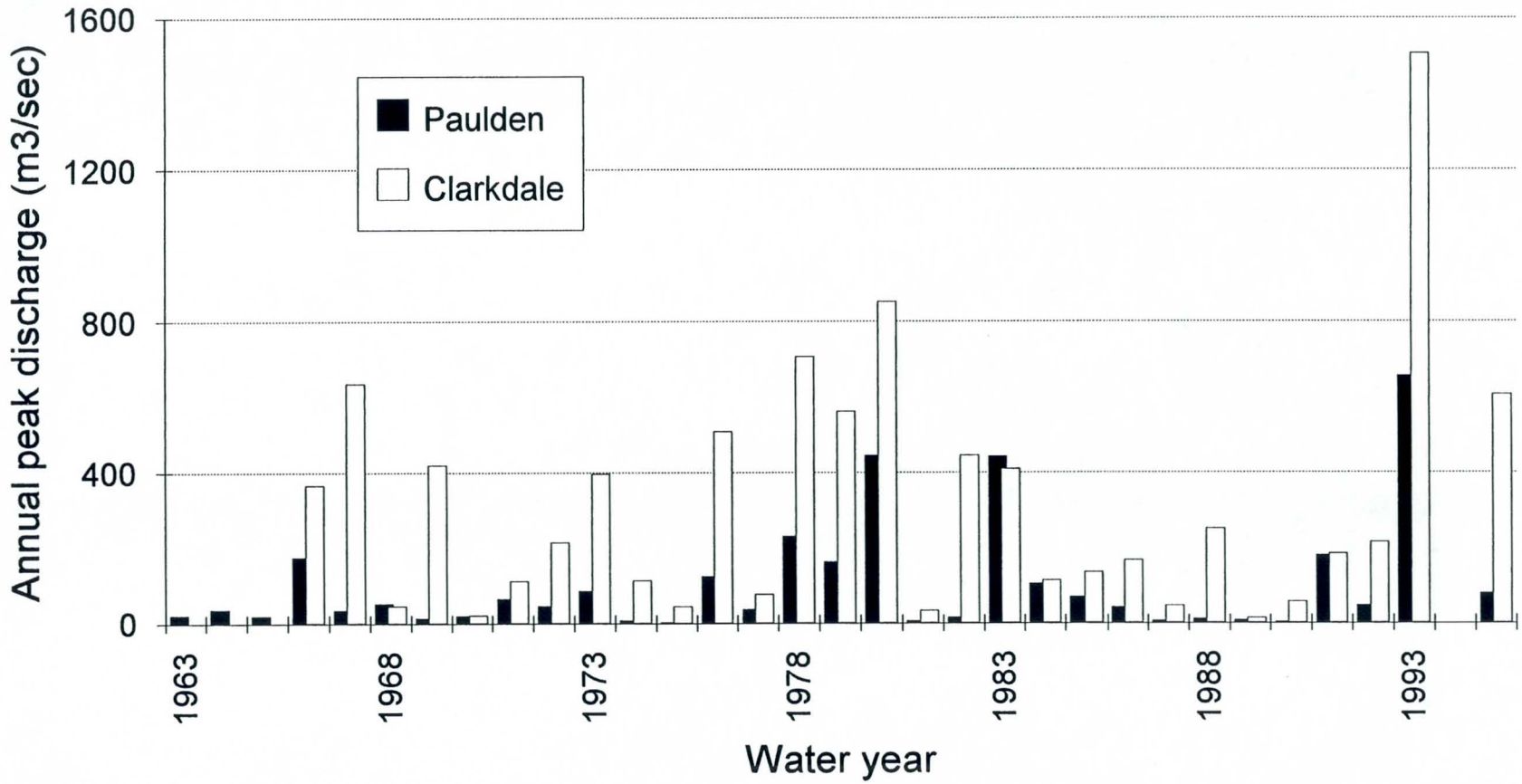

Figure 3. Annual peak discharge at Paulden and Clarkdale gages during water years 1963 to 1995 (U.S. Geological Survey records).

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INTERACTIONS OF PREDATION AND HYDROLOGY ON NATIVE SOUTHWESTERN FISHES: LITTLE COLORADO SPINEDACE IN NUTRIOSO CREEK, ARIZONA

John N. Rinne[1]

Non-native fish introductions in the southwestern United States have resulted in the reduction in and replacement of many native species. Interspecific interactions may come either singularly or as a combination of competition, hybridization, or predation (Rinne 1994). Competition between non-native, introduced species and native southwestern species is not easily or well documented (Douglas et al. 1994). The mechanism of hybridization is best documented for introduced salmonids and southwestern native trouts (Rinne 1985, 1988, 1994; Rinne and Minckley 1985). Predation by introduced species as a mechanism of interaction and impact on native species is increasingly being demonstrated as an important factor in the reduction and replacement of native fishes.

Minckley (1982) first suggested that the decline of the threatened razorback sucker, *Xyrauchen texanus*, resulted from predation on its eggs and fry by introduced species. Meffe (1985) demonstrated that the introduced western mosquitofish, *Gambusia affinis*, was an effective predator on the native endangered Gila topminnow, *Poeciliopsis occidentalis*. Marsh and Brooks (1989) reported the marked effect of ictalurid catfish predation on the young of the introduced razorback sucker. Blinn et al. (1993) reported that rainbow trout, *Oncorhynchus mykiss*, has a significant impact on the distribution and abundance of the native threatened Little Colorado spinedace, *Lepidomeda vittata vittata*. Rinne and Alexander (in press) examined further the role of salmonid predation on not only spinedace but the native Apache trout (*O. apache*). They concluded, based on laboratory and field studies, that predation by introduced rainbow trout and brown trout (*Salmo trutta*) is a major contributor to the decline in both numbers and range of these two native threatened species.

In addition to the biotic impacts of introduced species, habitat alteration and loss have contributed to the decline of native fish species in the Southwest (Rinne 1994). Habitat alteration has come largely through groundwater mining, damming, and diversion of streams and rivers. Many artificial lakes occur in upper elevation montane streams above the Mogollon Rim in Arizona. These impoundments are stocked with introduced salmonids for sport fishing. Rinne and Janish (1995) reported that over 69 million non-native fishes were stocked into the watersheds of the Little Colorado and Black rivers between 1930 and 1991. Ninety percent were stocked in lakes and reservoirs that are positioned in the headwaters of these two major river systems. Accordingly, they serve as a constant potential source of input for non-native fishes into downstream creeks and rivers. The objectives of this paper are to examine the relative roles of biotic (predation) and abiotic (water quality, hydrology, and its modification) factors in one montane stream in east-central Arizona, and also to expand results from Nutrioso Creek to river systems and other native species in general.

Study Area

Nutrioso Creek heads near 2700 m and flows north toward Springerville, Arizona (Figure 1). It is joined by Paddy Creek at about 2500 m and then passes through the town of Nutrioso. The reach above Nutrioso (ca. 7 km) has a higher gradient (> 5%) than the downstream reaches, with substrates mostly of cobble and boulder. From the confluence with Paddy Creek to several kilometers downstream it passes through a montane meadow. At base flows the waters are

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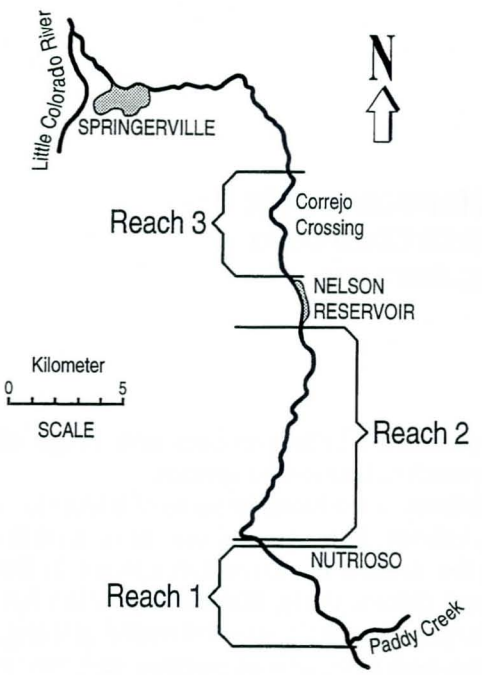

Figure 1. Map of Nutrioso Creek indicating reaches and Nelson Reservoir (modified from Blinn and Runck 1990).

clear (< 10 NTUs; HACH spectrophotometer) and cold (< 20°C) and contain adequate dissolved oxygen (7–9 mg/l; Blinn and Runck 1990). Below the town of Nutrioso the stream passes through another montane meadow that lies between Nutrioso and Nelson Reservoir about 12 km to the north (Figure 1). This reach is mostly privately owned and is subjected to grazing and agriculture. It is lower in gradient (2–5%) with gravel-sand to silt substrates. The stream channel is deeply incised (3–4 m) at many points and waters are diverted for irrigation (Marsh and Young 1988). Waters are only slightly warmer, dissolved oxygen is adequate, and turbidity is 20–25 NTUs. The third and final reach (Figure 1) lies below Nelson Reservoir, which was constructed in the 1950s. This reach also lies within montane meadow vegetation type, but the stream is characteristically deeply incised (1–3 m) into meadow substrates. Similar to Reach 2, its gradient is low (≤ 2%) and its substrate is primarily silt sand with pebble-cobble riffles. Turbidity is significantly higher, ranging from 100 (Rinne and Alexander in press) to over 250 NTUs (Blinn and Runck 1990) during the summer months. Oxygen can be reduced at night (< 2.0 mg/l), but it is adequate for both trout and spinedace during daylight hours (7 mg/l). Finally, Nelson Reservoir provides sport angling for residents of Springerville, Nutrioso and Alpine, Arizona and is periodically stocked with primarily rainbow trout. Between 1958 and 1991 over 3 million rainbow were stocked on 106 different occasions (Rinne and Janisch 1995).

Reach 1 contains almost exclusively rainbow x Apache trout hybrids with speckled (*Rhinichthys osculus*) being uncommon. Spinedace have only been reported in the lowermost portion of this reach at the town of Nutrioso (Highway 180/191 bridge; Marsh and Young 1988). Reach 2 contains rainbow, spinedace, two suckers (bluehead [*Catostomus discobolus*] and flannelmouth [*C. latipinnis*]), and another native cyprinid (speckled dace [*Rhinichthys osculus*]; Marsh and Young 1988). In addition to the above species, both brown and brook trout (*Salvelinus fontinalis*), which are introduced into Nelson Reservoir for sport fishing (Rinne and Janisch 1995), and fathead minnow (*Pimephales promelas*), which is a bait species, occur in Reach 3. In summary, there are four native and four introduced fish species occurring in Nutrioso Creek.

Materials and Methods

Hatchery-reared juveniles (60–100 mm, TL) of the native Apache trout, *Oncorhynchus apache*, were used in laboratory experiments. This species was used because it was readily available from the Williams Creek National Fish Hatchery (U.S. Fish and Wildlife Service). Further, potential predation effects on this species in the wild could be estimated. Finally, because of morphology and observed behavioral traits we used the Apache trout as a "surrogate" for the much rarer and difficult to obtain spinedace. Because introduced trouts and the native spinedace occur in the same reaches of the stream, a final objective was to determine the roles of turbidity and cover as factors in reducing introduced trout predation on the spinedace. Brown, rainbow, and hybrid (Apache x rainbow) trout were used as predators.

Predation relative to turbidity was conducted under field conditions in net enclosures constructed of 3.1 mm (1/8 inch) mesh and 1.5 m x 3.0 m in size. Nets were deployed in Nutrioso Creek (see Blinn et al. 1993) below Nelson Reservoir in summer for periods of 7 to 12 days. Mean turbidity (NTUs) was estimated from water samples taken at surface, mid-water, and bottom at three locations within each cage and analyzed

with a Hach turbidity meter. Spinedace (the prey) and trout (the predator) were measured, counted, and placed in the cages. At the conclusion of the experiment, all remaining fishes were removed and counted to determine the percent predation.

Laboratory studies were designed to permit controlled and elevated levels of turbidity in artificial, circular raceways (Frigid Units) 3–7 m in length. Fines (< 2 mm) used in studies were sieved from substrate materials collected in streams from the White Mountain region where all species occur in the wild. Water temperature was controlled by a Frigid Units chiller at 10–13°C. Water was circulated by a 1 HP pump to sustain turbidity. At the conclusion of the experiments, predator and prey were enumerated to permit calculation of percentage predation over the duration of the experiment. Additional laboratory studies were conducted in the presence of the abiotic influence of cover and with neither cover nor turbidity present. Cover comprised stream cobble (150–300 mm) placed in raceways. Percent cover was calculated as the percentage of surface area of the cobble relative to that of the total experimental area.

Laboratory Predation Study Results

Laboratory experiments indicated that two rainbow and brown trout could reduce Apache trout juveniles from 60 to 68 percent (mean = 60%) in a period of 24 hours (Table 1). Similar results occurred using brown trout as the predator; however, brook trout reduced Apache trout numbers by only 30 percent in days.

Similar experiments using brown trout as the predator with artificially high turbidities (>195 NTU) showed a somewhat reduced average loss of prey species (Apache trout; mean 60–45%) in a 3–4 day period (Table 2).

By comparison, artificially imposed 13–35 percent cover did not reduce the predation efficiency of either brown or rainbow trout (Table 3). In fact, mean loss (77%) was greater than in clear waters with no cover (mean 60%; Table 1).

Field Predation Study Results

Blinn et al. (1993) reported on the results of enclosure studies from Reach 1, the upper clearwater reach of Nutrioso. Rainbow trout (190–270 mm TL) in the presence of an adequate invertebrate food supply were effective and efficient predators on spinedace (40–65 mm TL). One 185 mm TL trout was reported to consume 4 spinedace in one 16-hour period. In addition, the presence of the rainbow predator affected the behavior of spinedace. Spinedace are trout like and utilize undercut banks for cover in absence of vegetation, boulders, or deeper water. In the absence of rainbow trout in enclosures, the spinedace used undercuts significantly more than in enclosures containing this non-native predator. Conversely, in the presence of rainbows, spinedace were forced into open waters,

Table 1. Results of laboratory predation studies using introduced salmonids (> 175 mm) as the predator species and hatchery-reared Apache trout (*Oncorhynchus apache*) as the prey.

Predator	No.	No. of Prey Start	No. of Prey End	% Loss	Duration (days)
Rainbow	2	25	9	64	1
Rainbow	2	25	8	68	1
Rainbow	2	25	10	60	1
Rainbow	2	25	10	60	1
Rainbow	2	25	9	64	1
Rainbow	2	25	8	68	1
Brown	2	25	10	60	1
Brown	2	25	10	60	1
Brook	2	10	6	30	3

Table 2. Results of laboratory predation studies with brown trout (*Salmo trutta*; > 200 mm) and Apache trout (*Oncorhynchus apache*; 75–100 mm) at varying artificially created turbidities.

No. of Predator	No. of Prey Start	No. of Prey End	% Loss	Duration (days)	Turbidity (NTUs)
4	40	15	66	4	214
4	40	30	25	3	200
2	20	6	70	3	195
2	45	35	22	4	200–300

Table 3. Results of laboratory predation studies using brown and rainbow trout (> 175 mm) as predators and Apache trout juveniles as prey and in varying percentages of cover.

Predator	No.	No. of Prey Start	No. of Prey End	% Loss	Duration (days)	% Cover
Rainbow	2	14	1	93	3	13
Rainbow	2	14	4	71	3	25
Brown	2	25	8	68	4	15
Brown	2	25	6	76	3	35

rendering them vulnerable to other visual predators such as great blue herons and garter snakes.

Field experiments using rainbow trout, hybrid Apache x rainbow trout, and brown trout in the presence of turbidity indicated that as turbidity increased, the percentage loss of prey decreased (Table 4). Compared to lab experiments conducted for about half the time period (4 vs. 8 days) and at lower turbidities (mean; 85 vs. 200 NTU), there was no difference in the comparative loss of prey species to predation. This assumes that there is no difference in prey response (Apache trout vs. spinedace) to the predator (brown trout).

Table 4. Results of field predation studies in cage enclosures using introduced salmonids (>175 mm, TL) and their hybrids as predators and Little Colorado spinedace (*Lipidomeda vittata vittata*; 40–60 mm, TL) as prey. Studies were conducted at naturally occurring turbidities in Nutrioso Creek on the Apache Sitgreaves National Forest.

Predator	No.	No. of Prey Start	No. of Prey End	% Loss	Duration (days)	Turbidity (NTUs)
Rainbow	0	40	27	33	10	no data
Rainbow	4	35	3	89	10	no data
Rainbow	4	35	1	97	10	no data
Rainbow	2	20	9	55	7	13
Rainbow	2	20	14	30	12	82
Rainbow	4	20	20	0	9	103
Hybrid	2	20	2	90	7	9
Hybrid	2	20	3	85	12	63
Hybrid	2	5	2	60	9	82
Brown	2	20	8	60	7	18
Brown	2	20	9	55	12	76
Brown	4	10	7	30	9	160

Discussion and Conclusions

Results of both field and laboratory predation studies indicated that predation as a mechanism of interaction with non-native trout is significant in the reduction and replacement of the threatened Little Colorado spinedace. Turbid waters exist below Nelson Reservoir in Nutrioso Creek where introduced trouts and spinedace co-occur. These turbidities may reduce predation effectiveness; however, potential loss to predation is yet significant (ca. 60% in a week to 10 days). Similarly, cover up to 35 percent in laboratory experiments did not effectively reduce predation efficiency of either rainbow or brown trout. Elevated turbidities may reduce predation but will not eliminate it. Indeed, sustained elevated turbidities is not a management option and may ultimately have negative effects on the native spinedace.

The presence of Nelson Reservoir and the continued stocking of salmonids for sport fishing is a major impediment to security of spinedace in Nutrioso Creek. Similar stocking in reservoirs in the East Clear Creek drainage further impacts the spinedace. Such stocking ensures a continued source of predators to the downstream, turbid, warmer, and often low dissolved oxygen reach where these salmonids would through time most likely be eliminated because of unsuccessful reproduction. Elevated turbidity and temperature below Nelson Reservoir have undoubtedly favored the spinedace more than the introduced trouts, thus sustaining populations of the native cyprinid. The middle reach (2) of Nutrioso is not optimum trout habitat. Even in the event of upstream migration of trout from Nelson Reservoir, they would not become established because of reduced habitat quality in the form of dewatering from irrigation and increased water temperature during summer months.

Stream renovation to remove non-native salmonids followed by reintroduction of native salmonids is a well-developed and often used technique in upper elevation streams (Rinne and Turner 1991). It might be suggested as an approach to remove introduced trout from the headwaters of Nutrioso Creek. Because the native Apache trout has occurred in the headwaters of Nutrioso Creek (Rinne 1985; Dowling and Childs 1992), it would be conceivable to reintroduce this species. The question is "Did the native trout and spinedace co-occur in the headwaters of this creek?" Relevant to this question are (a) whether spinedace occurred in the upstream, higher gradient colder waters, and (b) if native trout consume the spinedace as the introduced trouts do.

Preliminary experiments relative to the first question indicate that sustained spinedace populations never occurred in the upper portions of Reach 1. Two hundred spinedace were captured from Reach 3 and introduced to Reach 1 in October 1992. Rainbow trout were removed from a 200 m reach of the stream to remove the predation influence. In May of 1993 no spinedace were located in either the introduction pools or downstream for a half kilometer. An additional 50 adult spinedace (> 60 mm TL) were introduced on June 16 1993. Revisitation and sampling with electrofishing gear 5 weeks later revealed no

spinedace in the introduction pool. Only a half dozen were captured about 200 m downstream where movement was impaired by a hardware cloth barrier. Beaver dams in this reach historically may have permitted spinedace to penetrate further into the headwaters of Nutrioso. Presently, they are found only up as far as the town of Nutrioso where beavers persist today.

With regard to predation by native trout, laboratory experiments indicate that Apache trout smaller than 225 mm TL do not prey upon spinedace. Two dozen spinedace (40–60 mm TL) were placed in a living stream (Frigid Units) with a dozen wild caught Apache trout (90 to 225 mm) for a period of 2 weeks. No spinedace were lost. However, when two large (400 mm TL) hatchery-reared Apache trout were similarly placed in this same environment with two dozen spinedace, most were consumed within an hour. Because the maximum size of Apache trout is limited by physical habitat to 250 mm or less in upper Nutrioso Creek, predation by the native if spinedace co-occurred with it would probably be minimal.

Altered stream hydrology through dams, which provides habitat for introduced sport fishes in situ and alters flow regimes ex situ or downstream, is a major contributing factor to the decline of the Little Colorado spinedace. As such, dams present both an abiotic and biotic impact on the spinedace and other native fishes. Many other species of native fishes depend on natural hydrographs for their life history requirements (Meffe and Minckley 1987; Rinne and Minckley 1991). Disturbance events such as floods stimulate native fishes to spawn (Mueller 1984) while at the same time reducing or removing non-native species (Meffe and Minckley 1987; Stefferud and Rinne in press). On the other hand, dams alter, control, and stabilize flow regimes, rendering them more favorable habitat for many introduced species, primarily from the Mississippi River Basin.

Co-occurrence of native and non-native fishes in the same reaches of a stream or river greatly reduces the chances of native species surviving through time. If natural hydrographs are present, then the periodic, abrupt flooding characteristic of the Southwest streams may favor sustainability of the natives. In contrast, controlled streams below lakes and reservoirs ameliorate these events and lower the probability of sustaining native fish faunas. The upper Verde and Gila rivers may yet sustain native fish faunas because they are not dammed (Propst et al. 1986, 1987; Stefferud and Rinne 1995).

Ideally, exclusion of non-natives is necessary if native species are to persist. This approach has been used widely in upper elevation montane streams such as Nutrioso Creek and many others in the White Mountains of east-central Arizona (Rinne and Turner 1991) and continues to the present (Rinne and Janisch 1995). In larger lower elevation streams in the Southwest logistics prevent this approach. One approach to sustaining native fishes is securing habitats for them by purchase. This has been done for the spinedace through purchase of real estate encompassing tributaries and the mainstream Little Colorado River (Rinne and Janisch 1995). Watersheds can be managed as special areas for native species.

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EFFECTS OF PRESCRIBED FIRE ON WATERSHED RESOURCES: A CONCEPTUAL MODEL

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The occurrence of fire in southeastern Arizona ecosystems has been documented since the middle of the nineteenth century. Bahre (1985) concluded that wildfires occurred frequently in all major vegetation communities in southeastern Arizona, including desert shrub, between 1859 and 1890, and that lower frequencies after that period played an important role in the "brush invasion" that started in southeastern Arizona in the 1890s. Wildfire frequency decreased substantially in grasslands after 1882, probably because of overgrazing and the early efforts of Anglo settlers to suppress fire. The suppression of fire and subsequent invasion of brush was reported to be contemporaneous also with increased erosion in the southwestern United States (Leopold 1924).

Several authors have prepared comprehensive reviews on prescribed burning in Arizona ponderosa pine forests (Biswell et al. 1973; Wright 1978) and in mixed conifer, ponderosa pine, pinyon–juniper, and chaparral (Arnold 1963; Zwolinski and Ehrenreich 1967). Despite information on the use of prescribed fire in ecosystem management presented by earlier investigators (Cooper 1961; Kallender 1969; Lindemuth 1960; Weaver 1951) and in more recent studies (Covington and Moore 1992; Swetman 1990), the implementation of prescribed burning programs has been slow in many areas. As a result, critical fire hazard conditions have continued to develop as fuel loading has increased over the years of fire exclusion. For this reason, it is important to revisit the information available on prescribed fire and to place this knowledge in a current perspective, particularly with regard to the impact of different fire severities on soil and water resources.

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Substantial scientific information on fire history, vegetation responses, fire behavior, nutrient cycling, and fire prescriptions has been published for a range of southwestern conditions and vegetation types. Some of this information was compiled and published in an earlier proceedings for a symposium on fire ecology and the control and use of fire in wild land management (Wagle 1969). A more recent, and comprehensive, synthesis of fire effects information is contained in the *Proceedings of a Symposium on Effects of Fire Management of Southwestern Natural Resources* (Krammes 1990). The symposium was held in Tucson, AZ in 1988. Discussion of the effects of fire on watershed and soil was included at this symposium. Although there is a large body of published information on the effects of fire on watershed and soil, much of this literature reports the effects of wildfires. Consequently, little quantitative information is available on the effects of prescribed fires on runoff/infiltration, sedimentation, and nutrient losses by runoff and erosion (Robichaud and Waldrop 1994; Robichaud et al. 1993). This is particularly true for watersheds in the southwestern United States.

There are several pragmatic reasons for the abundance of information on wildfire effects. First, the cost of wildfire assessment and associated data collection is usually included as part of emergency funding associated with wildfire suppression and rehabilitation—for example, the cost of monitoring during emergency rehabilitation treatments following fire. Funding at this level is not usually available for prescribed burning and fuels reduction programs. Changes resulting from wildfires are often much easier to document and validate than the more subtle changes produced by lower intensity prescribed fires. As a result, much of the published literature emphasizes severe wildfires that have significant effects on the watershed and associated

soil and water resources. However, there are isolated reports throughout the literature that report on watershed responses associated with less severe prescribed fires.

Even in the case of wildfires, effects will depend on factors such as postfire precipitation patterns and management activities, and thus prediction of effects becomes a complex process. For example, it has been extremely difficult to quantitatively establish the relationship between the establishment of ryegrass cover and the reduction in erosion immediately following fire because of geographic and annual climatic variability (Barro and Conard 1987).

It is the objective of this paper to develop a simple conceptual model relating fire severity to watershed responses. It can be used as an initial framework for portraying information on watershed and soil responses that have been reported for vegetation and climatic conditions in the southwestern United States.

The Conceptual Model

It is important when discussing fire effects on soil and water to clearly differentiate between fire intensity and fire severity. Fire intensity is a term understood by fire behavior specialists to be the rate of energy release per unit of ground surface area and is proportional to flame height and rate of spread (Chandler et al. 1983). Because fire intensity measurements are difficult to relate to specific soil and water responses (Hungerford 1989), fire severity has been used to describe the amount of vegetation and soil changes associated with a particular fire (Wells et al. 1979). The relationship between fire intensity and fire severity for different vegetation types remains largely unsolved, although substantial progress is being made in developing quantitative models to describe changes in thermal conductivity in soils (Campbell et al. 1994) and soil temperature and water content beneath surface fires (Campbell et al. 1995). These relationships are then being used to develop models that describe fire-driven heat and moisture transport in soils (Albini, in preparation). However, because these quantitative models have not been fully implemented, the resource responses discussed in this paper will refer primarily to different levels of fire severity.

The conceptual model described below portrays fire severity as a continuum ranging from minor resource responses under a cool-burning prescribed fire to major responses that could be expected to occur during stand-replacing wildfires in forests. The fire response continuum in the southwestern United States is large (Baker 1990). In Figure 1, prescribed fire conditions are depicted on the left side of the fire response continuum and represent lower temperature–higher humidity burning conditions where fuel loading is minimal and fuel moisture is high. These conditions produce lower fire intensities, and thereby expose the soil and water resources to lower fire severities. Prescribed fire usually has minor hydrologic impacts on watersheds because the surface vegetation, litter, and forest floor are only partially burned (Baker 1990). Other resources (soils, wildlife, vegetation) are also changed very little by a prescribed fire. On the other end of the fire response continuum (right side of Figure 1), fire behavior more nearly represents that present during wildfires, where the temperatures, wind speeds, and fuel loadings are high, and the humidities and fuel moisture are low. In contrast to prescribed burning, wildfires can have a major effect on basic hydrologic processes, leading to increased sensitivity of the site to eroding forces and to reduced land stability (Baker 1990). Large changes also occur in the other resources (denuded landscapes, large losses of plant nutrients, and so on).

The differences in impacts between prescribed burning and wildfires depend partly on the vegetation type being burned. For example, in ponderosa pine forests there can be large differences; during a cool prescribed fire, only the litter and smaller diameter surface fuels are ignited as compared to near total canopy consumption during intense wildfire. In contrast, fires burning in chaparral are mainly carried by the shrub canopies. Therefore, it is more difficult to control the behavior and intensity of the fire so that only minimum impacts to the soil and vegetation occur. In order to obtain less severe fires in chaparral, fires are ignited during marginal burning conditions, or by using special heat-generating ignition techniques (e.g., helitorch). Also, low-severity fires in chaparral often result in mosaics of burned and unburned patches because the slight differences in slope and aspect make total ignition and coverage impractical.

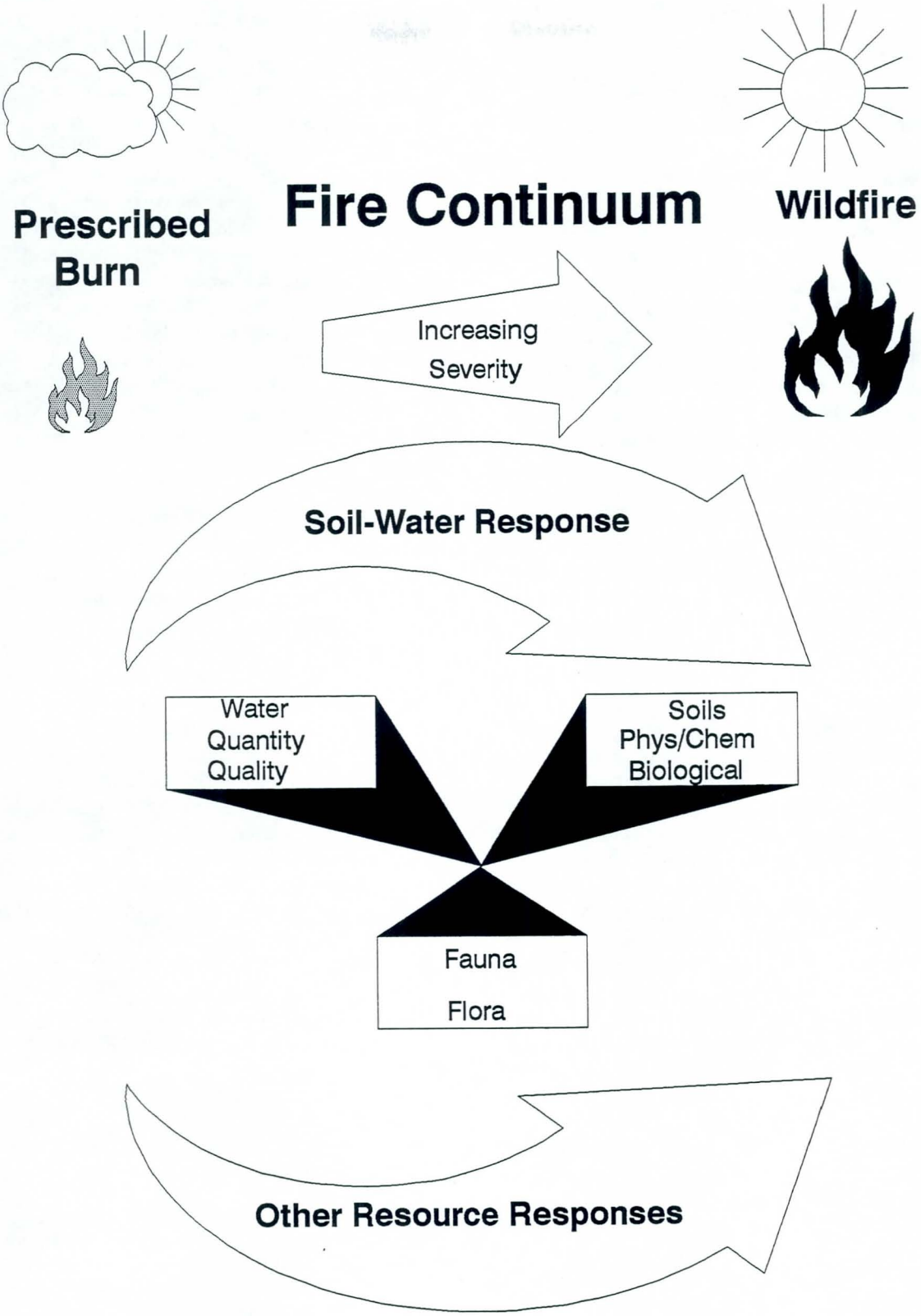

Figure 1. Conceptual model relating immediate resource responses to a fire severity continuum extending from cool-burning prescribed fires to severe wildfires.

Data for Model Development

Before the conceptual model described above can be developed, the response functions for the different resources to a range of fire severities (extending from low to high severity fires) need to be defined. Another important factor affecting the postfire hydrologic scenario is the postfire precipitation pattern. The hydrologic response model is visualized as being a three-dimensional surface with a particular hydrologic response (peak flow, sedimentation rate, etc.) as being a function of both fire severity and a time variable reflecting climatic events following a fire. The immediate soil and watershed response would be most closely related to fire severity (e.g. how much litter and plant cover had been destroyed by the fire, amount of nitrogen volatilized, etc.), and would probably be a nonlinear function, as the one that describes infiltration into a wettable dry soil. An additional time function, reflecting precipitation events, would be necessary to define the longer term hydrologic responses to a particular fire severity. The dimension of time is essential for the model because of the possibility of variable precipitation events that could follow a fire. For example, a low-intensity prescribed fire can produce substantial runoff and soil loss as sediment if the fire is immediately followed by high-intensity rainstorm events. Conversely, severely burned watersheds can produce little runoff and erosion if a fire is followed by a relatively mild year with gentle rains and warm temperatures that allow a protective vegetation cover to develop. The time function that reflects precipitation events will be stochastic in nature and will have to be constructed within a probability framework so that best estimates of outcomes can be determined. Information for the two-dimensional part of this model has been developed by Ffolliott et al. (1988), and with some modification could be used as a starting point for developing the time dimension following fire.

The first iteration of the above model is being prepared to describe soil and watershed responses. All available information on hydrologic responses for Arizona and the southwestern United States is currently being consolidated in order to define and quantify hydrologic responses to both wildfires and, more importantly, cooler burning prescribed fires. Unfortunately a very meager data base is available on the hydrologic responses of watersheds to lower fire severities. The best hydrologic response data available in the literature are for ponderosa pine forests (Campbell et al. 1977; Gottfried and DeBano 1990; Rich 1962; Sims et al. 1981; Zwolinski 1971) and Arizona chaparral (Davis 1989; Glendening and Pase 1961; Heede 1990; Hibbert et al. 1974; Pase and Ingebo 1965; Pase and Lindemuth 1971). Soil response data is likewise most readily available for ponderosa pine (Covington and Sackett 1984, 1990; Wagle and Eakle 1979) and Arizona chaparral (DeBano 1989, 1990; Weinhold and Klemmedson 1992). Much less information exists for watershed and soil responses to fire in mixed conifer, pinyon–juniper, and desert grasslands, although there is pertinent information in nearby areas in the western United States that are applicable to Arizona and the southwestern United States.

Concluding Statement

Prescribed fire continues to be an important tool for managing southwestern ecosystems. The use of fire, however, must be carefully planned and implemented in order to gain the desired response without damaging the watershed resources. It is important when planning fires to clearly differentiate between prescribed burning and wildfires and to burn under cooler conditions. A conceptual model has been developed to illustrate more clearly the differences in impacts between prescribed burning and wildfires on different watershed resources. Although there is much information on the fire impacts on watershed resources, most of it has been collected after wildfires. Scant information is available on the soil and water resource responses to lower severity prescribed fires. Any model describing hydrologic responses must also include a time dimension that represents the sequence of postfire precipitation events.

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EFFECTS OF FIRE ON WATER RESOURCES—A REVIEW

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Fire exerts a tremendous influence over forest ecosystems in North America depending on its intensity, duration, and frequency. It is an important natural disturbance that has played a significant role in the development of most forest ecosystems. The "natural" fire that shaped these ecosystems occurred across a continuum that ranged from light burn to catastrophic conflagration. Thus, fire disturbance to riparian and aquatic ecological systems within these forests produces a continuum of effects on water resources.

During most of the twentieth century, foresters focused a lot of attention relative to fire on the multiple resource damages produced by wildfire. Indeed, catastrophic fires produced by natural events (drought, insect outbreaks, lightning) did result in serious impairment to water resources. In the past decade, forest land managers have gained a great understanding of fire's significance in forest ecosystems by observing both the catastrophic effects of decades of an "unnatural" disturbance (fire exclusion), as well as the beneficial effects of well-managed prescribed fire programs.

As a physical–chemical process, fire is a continuum that results from interactions of intensity, climate, slope, topography, soils, and area. Thus the impact on water resources also occurs on a continuum. Our ability to state and describe these impacts is a function of the scientific information obtained from a limited range of fires. Obviously, fires and water resource impacts will continue to occur on new and unique combinations across this continuum.

Fire intensity refers to the rate at which a fire is producing thermal energy. The higher the intensity, the more severe the impacts on water resources. Intensity is a function of climate, temperature, rate of spread, heat yield, and fuels. Temperatures can range from 50 to 1,000°C. Rates of spread can vary from 1 m in 2 weeks (peat fire) to 6–7 km/hr (large wildfire). Heat yields range from 2 to 2,000 BTU/kg, and fuels grade from grasses (1 Mg/ha) to heavy timber (160 Mg/ha). Climate, slope, topography, soils, and watershed size are other continuums that act to affect fire intensity.

Watersheds are used as the basic unit of measure for ecosystem analysis since water is the main transport mechanism that integrates ecosystem processes. Watersheds function on all time and spatial scales. They are also a focus for important human activities (water supply, recreation, resource production). An important part of understanding the impacts of fire on water resources is to comprehend the processes involved. A lot of information has been incorporated here to describe the range of impacts. Since fire is a continuum and data are scarce for some portions of this continuum, not all situations will be adequately described. Therefore, understanding of the processes is a key factor in successful interpretation of the effects of prescribed fires and wildfires on water resources.

Watershed condition, or the ability of a watershed system to receive and process precipitation without ecosystem degradation, is a good predictor of the potential impacts of fire on water resources. The surface cover of a watershed consists of organic forest floor (thin to thick), vegetation (variable cover), bare soil, or rock. Fire can destroy the organic forest floor and vegetation, and alter the infiltration and percolation capacity of bare soil. In some soil-vegetation complexes, water repellency can develop and greatly reduce water infiltration

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(DeBano 1971). This alters watershed (hydrologic) condition, with erosion increasing as watershed condition goes from good to poor.

Water Quantity

Fires affect the quantity of water in a forest ecosystem by affecting key water cycle processes. Fires can reduce interception, thereby reducing moisture storage, increasing water yields, and creating greater runoff with smaller storms. Burning the forest floor reduces litter storage of precipitation (0.5 mm/cm of litter lost) and increases overland flow. Fires temporarily eliminate transpiration, increasing soil moisture and streamflow. Burning off surface organic matter reduces infiltration, thereby increasing overland flow and surface stormflow. Fires usually increase streamflow in most forest ecosystems, but can decrease streamflow in snow-dominated and fog-drip systems. Fires alter baseflow and increase stormflow volume and response. Watershed response to storm events is greater with shortened time to peakflow and susceptibility to flash floods is greater. Fires increase snow accumulation and melt in burns of less than 4 ha, but reduce snowpacks where burns exceed 4 ha.

Total Yields

The effects of disturbances, primarily harvesting, on water yield from forested watershed studies throughout the world have been well documented and reviewed (Anderson et al. 1976; Bosch and Hewlett 1982; Neary and Hornbeck 1994). For the most part, water yields increase when mature forests are harvested, burned, blown down, or attacked by insects. The only exceptions occur where fog is abundant or snowfall accounts for a majority of the annual precipitation. The magnitude of measured water yield increases the first year after a fire disturbance and can vary greatly at one location or between locations depending on fire intensity, climate, precipitation, geology, soils, watershed aspect, tree species, and proportion of the forest vegetation burned. Since measured increases in water yield are primarily due to elimination of plant cover, with subsequent reductions in the transpiration component of ET, yield increases have been found to be greater in ecosystems with high ET. Streamflow increases produced by forest disturbances decline as both woody and herbaceous vegetation regrow. This recovery period can range from a few years to decades.

Most of the water yield information deals with the effects of harvesting. Water yield can increase 0–126 percent from harvesting (Neary and Hornbeck 1994). Water yield increases from prescribed fires and wildfires are shown in Table 1. Increases in water yield are highly variable, but generally are greater in regions with high ET.

Baseflows

Baseflows are important in maintaining perennial flow through the year. They are critical for aquatic species (no water, no life). Baseflows can increase if the watershed condition remains good (when infiltration rates are adequate). If the watershed condition deteriorates and more precipitation leaves as surface runoff, baseflows will decrease. In extreme conditions, perennial streams become ephemeral.

Surface runoff (overland flow) occurs when the rain intensity or snowmelt rate exceeds infiltration capacity. It frequently occurs on rock, compacted soils, shallow soils, water repellent soils (after fires), and roads. Overland flow that is 1 percent of rainfall in undisturbed forests can increase to 15–40 percent after fires.

Stormflows, Peak Discharges, and Response Times

The effects of forest disturbance on storm peakflows are highly variable and complex. They can produce some of the most profound impacts that forest managers have to consider. Anderson et al. (1976) offer a good review of peakflow response to disturbance. Peakflows after forest cutting can decrease 66 percent or increase 100 percent depending on location, the percent of the watershed cut, and season (Table 2). Most studies show increases in peakflows of 9–100 percent. The study with an average of 100 percent after clearcutting ranged from -19 to +250 percent for individual storms. An analysis for the Pacific Northwest indicated that if 1 percent of large watersheds were clearcut, peakflows for 100-year floods could increase 6 percent, and annual floods would increase 20 percent. There has been some concern that increases in annual flood peaks of 20+ percent could lead to channel instability and degradation.

Fire has similar to larger effects on peakflows. The Tillamook Burn in 1933 in Oregon increased the total annual flow of two watersheds by 9

Table 1. Increased Water Yield from Burned Watersheds.

Location	WS Condition	Area (ha)	PPT (mm)	Runoff (mm/yr)	Recovery (yrs)	References
S. Carolina	Loblolly Pine		1390			Van Lear, Douglass,
	Control	2		124		Cox and Augspurger 1985
	Understory Burn	2		180	2	
	Burn + Harvest	2		217	2	
Texas	Juniper\Grass		660			Wright, Churchill and
	Control	<1		2		Stevens 1982
	Burned	<1		25	5	
	Burned, Seeded	<1		10	2	
Arizona	Chaparral		740			Davis 1984
	Control	28		64		
	Burned	33		156	>11	
Arizona	Chaparral		585			Hibbert, Davis and
	Control	39		82		Knipe 1982
	Wildfire	39		130	?	
Arizona	Chaparral		655			Hibbert 1971
	Control	19		0		
	Wildfire	19		124	9+	
	Control	39		19		
	Wildfire	39		289	9+	
Arizona	Pinyon-Juniper		480			Hibbert, Davis and
	Control	5		34		Knipe 1982
	Burned	5		39	5	
	Control	5		43		
	Herbicide	5		56	5+	
Arizona	Pinyon-Juniper		482			Clary, Baker, O'Connell,
	Control	134		20		Johnson, Jr., and
	Slash Burned	134		11	4	Campbell 1974
	Control	147		18		
	Herbicide	147		28	4+	
Washington	Pine & Doug-fir		580			Helvey 1980
	Control (preburn)	517		221		
	Wildfire (postburn)	517		314		

percent and increased the annual peakflow by 45 percent. A 127 ha wildfire in Arizona increased summer peakflows by 500–1500 percent, but had no effect on winter peakflows. Another wildfire in Arizona produce a peakflow 58 times greater than an unburned watershed during record autumn rainfalls. Watersheds in the Southwest are much more prone to these enormous peakflow responses due to climatic and soil conditions. Studies have shown both increases (+35%) and decreases (-50%) in snowmelt peakflows.

Another concern is the timing of stormflows or response time. Burned watersheds respond to rainfall faster, producing more flash floods. Hydrophobic and bare soils, and cover loss will cause flood peaks to arrive faster and at higher levels. Flood warning times are reduced by "flashy" flow and higher flood levels can be devastating to property and human life. As indicated in Table 2, the Southwest is particularly vulnerable to changes in peakflow response time and volume. Another aspect of this is the fact that recovery times can range from years to many decades.

Table 2. Effects of cutting and fire on peak flows (from Anderson et al. 1976).

Location	Treatment	Peakflow Change (%)	
N. Carolina	Clearcut	+9	
W. Virginia	86% Cut	+21	
	Clearcut	+100	
New Hampshire	Clearcut	+100	(-19 to +250%)
New Hampshire	Clearcut	+35	(Snowmelt)
		-66	(Summer)
Oregon	Clearcut	+90	(Fall storms)
		+28	(Winter storms)
Oregon	Clearcut, Burned	+30	
	Clearcut, Burned (50%)	+11	
Oregon	Wildfire	+45	
Arizona	Wildfire	+500	(Summer Flows)
		+1500	(Summer Flows)
		0	(Winter Flows)
Arizona	Wildfire	+5800	(Fall storms)
Colorado	Clearcut, Burned	-50	
N. Carolina	Prescribed Fire	0	

Water Quality

Erosion

The main features of fires that affect erosion are wildfires (more intense and of greater area than prescribed fires), fireline construction, temporary roads, watershed rehabilitation activities, and increased storm peakflows. Soil loss can take the form of sheet, rill, or gully erosion. Fire-associated debris avalanches are a form of mass wasting that delivers sediment directly to streams in large quantities.

Rotational slumps close to channels can also be sources of sediment. These are mostly associated with water repellency conditions. Chaparral vegetation in the Southwest constitutes a high hazard for debris avalanches.

A stable stream channel reflects a dynamic equilibrium between incoming and outgoing sediment. Increased peakflows after fires can alter this equilibrium by transporting additional sediment into channels (aggradation) and by increasing peakflows that result in channel erosion (degradation).

Sediment Yields

Sediment yields from prescribed burns and wildfires are shown in Tables 3a and 3b. Wildfires generally produce higher sediment yields than other types of disturbances. Slope definitely aggravates sediment losses. After fires, turbidity can increase due to the suspension of ash and soil particles from silt to clay sized in streamflow. Turbidity is an important water quality parameter since high turbidity reduces municipal water quality, and can adversely affect fish and other aquatic organisms. Extra coarse sediments (sand, gravel, boulders) transported off of burned areas or as a result of increased storm peakflows can adversely affect aquatic habitat, recreation areas, and reservoirs. Deposition of coarse sediments destroys habitat and fills in lakes or reservoirs.

Anions–Cations

Undisturbed forests usually have tight cycles for major cations and anions, resulting in low concentrations in streams. Disturbances such as cutting, fires, and insect outbreaks interrupt or terminate uptake by vegetation and speed up mineral weathering, element mineralization, microbial activity, nitrification, and decomposition. These processes result in the increased concentration of inorganic ions in soil solution and leaching to streams via subsurface flow.

Nutrients carried to streams can increase the growth of aquatic plants, reduce the potability of water supplies, and produce toxic effects. Anions like phosphate and cations such as calcium and potassium can be exported from watersheds at 10 times their normal rate immediately after severe disturbances, but don't significantly alter water quality.

Most of the attention relative to water quality after fires focuses on nitrate nitrogen (NO_3-N) because it is highly mobile. Concentrations exceeding 10 mg/L (water quality standard) can cause methemoglobanemia in infants. High NO_3-N levels in conjunction with phosphorus can also cause eutrophication of lakes and streams. Most studies of forest disturbances show increases in NO_3-N (Table 4).

Fire Retardant

Ammonium-based fire retardants (diammonium phosphate, monoammonium phosphate, ammonium sulfate, or ammonium polyphos-

Table 3a. Sediment losses the first year after prescribed burns and wildfires.

Location	WS Condition	PPT (mm/yr)	Sediment Loss 1st Year (Mg/ha)	References
Texas	Juniper\Grass	660		Wright, Churchill and Stevens 1982
	Control		0.060	
	Burned (R = 3 yr)		15.000	
	Burned, Seeded (R = 1 yr)		3.000	
Montana	Larch, Douglas-fir			Debyle and Packer 1972
	Control		<0.001	
	Slash Burned		0.150	
Arizona	Chaparral			Glendening, Pase and Ingebo 1961
	Control		0.175	
	Wildfire		204.000	
California	Ponderosa Pine			Biswell and Schultz 1965
	Control		<0.001	
	Understory Burn		<0.001	
California	Chaparral			Clary, Baker, O'Connell, Johnsen, Jr., and Campbell 1974
	Control		5.530	
	Wildfire		55.300	
Arizona	Ponderosa Pine			Campbell, Baker and Pfolliott 1977
	Control		0.003	
	Wildfire		1.254	
Mississippi	Scrub Oak	1620		Meginnis 1935
	Control		0.056	
	Burned		0.739	
Oklahoma	Mixed Hardwoods	777		Daniel, Elwell and Cox 1943
	Control		0.022	
	Annual Burning		0.246	
N. Carolina	Southern Hardwoods	1190		Copley 1944
	Control		0.004	
	Semi-annual Burn		6.899	
Texas	Loblolly Pine	1040		Pope, Archer, Johnson et al. 1946
	Control		0.112	
	Annual Burning		0.806	
Texas	Loblolly Pine	1040		Ferguson 1957
	Control		0.224	
	Single Burn		0.470	

Table 3b. Sediment losses the first year after fires.

Location	WS Condition	PPT (mm/yr)	Sediment Loss 1st Year (Mg/ha)	References
Mississippi	Scrub Oak	1650		Ursic 1970
	Control		0.470	
	Burned (R = 3 yr)		1.142	
California	Chaparral	950		Wells, II 1981
	Control		0.043	
	Wildfire (R = 3 yr)		28.605	
Arizona	Chaparral	635		Pase and Lindenmuth, Jr. 1971
	Control		0.000	
	Control Burn		3.778	
Arizona	Chaparral	585		Pase and Ingebo 1965
	Control		0.096	
	Wildfire (R = 4 yr)		28.694	
	Wildfire, grass/ herbicide (R = 4)		66.151	
Arizona	Mixed Conifer	635		Hendricks and Johnson 1944
	Control		<0.001	
	Wildfire, 43% slope		71.680	
	Wildfire, 66% Slope		201.600	
	Wildfire, 78% Slope		369.600	
S. Carolina	Loblolly Pine	1390		Van Lear, Douglass, Cox and Augspurger 1985
	Control		0.027	
	Understory Burn (R = 2)		0.042	
	Burn, Cut (R = 2)		0.151	
Arkansas	Shortleaf Pine	1317		Miller, Beasley and Lawson 1988
	Control		0.036	
	Cut, Slash Burn		0.237	
Washington	Mixed Conifer	1475		Helvey 1980
	Control		0.028	
	Wildfire		2.353	
New Zealand	Native Podocarps	2610		O'Loughlin, Rowe and Pearce 1980
	Control		0.429	
	Cut, 20m Buffer, Burn		0.611	
	Cut, No Buffer, Burn		3.432	

Table 4. Effect of forest disturbances on maximum NO_3-N levels in streamflow (from Neary and Hornbeck 1994; and Neary and Michael in press).

Location	Forest Type	Treatment	Maximum NO_3-N (mg/L)
1. Cutting			
New Hampshire	Hardwoods	Clearcut	6.1
West Virginia	Hardwoods	Clearcut	1.4
North Carolina	Hardwoods	Clearcut	0.2
Oregon	Douglas-fir	Clearcut	2.1
2. Herbicides			
New Hampshire	Hardwoods	Cut, Herbicide	17.8
North Carolina	Hardwoods	Cut, Herbicide	0.7
Georgia	Pine\Hrdwds	Herbicide, Cut	5.3
Arizona	Chaparral	Herbicide	15.3
3. Fires			
Oregon	Douglas-fir	Cut, Burn	0.6
Arizona	Chaparral	Herbicide, Burn	18.4
Arizona	Chaparral	Prescribed Fire	12.0

Table 5. Temperature increases resulting from cutting and fire.

Location and Treatment	Buffer Strip	Temperature Increase (°F)	Reference
Oregon: Clearcut	Yes: 15-30 m	14m	Pase and Ingebo 1965
Pennsylvania: Clearcut	Yes: 30 m	3m	Pase and Lindenmuth, Jr. 1971
Oregon: Patch Cut	Yes: ?? m	0	Pope, Archer, Johnson et al. 1946
Clearcut	No	30d	
Oregon: Clearcut, Burn	No	13sm	Ursic 1970
Washington: Wildfire	No	10sm	Van Lear, Douglass, Cox and Augspurger 1985

sm = summer mean temperature
m = mean temperature
d = daily temperature

phate) play an important role in protecting forest resources from destructive wildfires. However their use can affect water quality, producing short-term mortality in some aquatic organisms. For aquatic organisms there is a tradeoff that needs to be considered: Fire and heat can be more destructive to aquatic resources in both the short and long term.

The main chemical of concern in streams 24 hrs after a retardant drop is ammonia nitrogen (NH_3^+ NH_4^+). Non-ionized ammonia (NH_3) is the principal toxic component to aquatic species. The distance downstream at which potentially toxic conditions persist depends on stream volume, the number of retardant drops, and the orientation of drops to the stream's long axis. Concentrations of $NH_3 + NH_4^+$ can reach 200–300 mg/L within 50–100 m below drop points. Under the right concentrations, toxic levels may persist for over 1,000 m of stream channel.

Light and Temperature Effects of Fire on Aquatic Habitat and Biota

Large fires can function like clearcuts in raising stream temperatures due to the direct heating of the water surface; increases of 0–30°F have been measured (Table 5). The two main concerns are reduction in the concentrations of dissolved oxygen and increased aquatic plant growth.

The recruitment of coarse woody debris into streams usually increases immediately after wildfires. This has a positive effect on fish habitat in the short term, but long-term inputs can be disruptive. Prescribed fires generally do not significantly affect coarse woody debris dynamics.

Fish Spawning Habitat

The main impact here is with sediment deposition in spawning gravels. Fine sediments released by fires (primarily wildfires) can clog interstitial spaces and reduce hatching success. Large-scale shifting of bedloads can also impact fish habitat.

Summary and Conclusions

Fires in forest and range ecosystems have a wide range of effects depending on the intensity and resultant hydrologic events. Wildfires definitely produce the largest effects, as they tend to be more intense and cover larger areas. The Southwest has recorded some of the largest changes in streamflow, peakflow, total water yield, and water quality due to the steep topography, the intense precipitation events of the region, and the nature of its soils. Although a lot of information exists about the effects of fires on water resources, efforts need to be made to put this information into a systematic context that can be used for wildland management purposes.

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PRELIMINARY OBSERVATIONS ON THE TRANSPORTATION OF LARGE WOODY ORGANIC DEBRIS IN BURNED AND UNBURNED HEADWATER STREAMS, TONTO NATIONAL FOREST, ARIZONA

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There are no published data on the transportation or accumulation of large woody debris (LWD) within aquatic ecosystems of the southwestern United States. The information available on LWD in the Southwest includes a review of the role and reduction of LWD in "hot desert" streams in the Southwest from a historical perspective (Minckley and Rinne 1985). Heede (1985) examined channel adjustments to the removal of log steps in two streams in the White Mountains of Arizona. In Wyoming and Colorado where climates are more similar to the Southwest than the Pacific northwestern states, Young (1994) examined the movement and characteristics of stream-borne coarse woody debris in adjacent burned and undisturbed watersheds in Wyoming. Richmond and Fausch (in press) studied the characteristics and function of large woody debris in mountain streams of northern Colorado.

Most of the published literature available on LWD comes from the Pacific Northwest and eastern United States. Information on LWD accumulation and transport, however, is not directly comparable with the Southwest because of meteorological, hydrological, and vegetation differences. Studies have examined the role of LWD in the following ways: (a) creating and providing fish habitat, protection from predation, and rearing areas for under-yearling salmonids (Bryant 1983; Bisson et al. 1982; Andrus et al. 1988; Murphy et al. 1984; Angermeier and Karr 1984); (b) storing sediment within a stream system (Klein et al. 1987); (c) providing stability or causing instability in the bed and banks of channels (Keller and Swanson 1979; Smith et al. 1993; Cherry and Beschta 1989); (d) as a source for decomposition products supplying the stream system with nutrients (Bilby and Likens 1980; Bilby 1981); and (e) as a food source and habitat for aquatic invertebrates and microorganisms (Cummins 1974, 1975; Sedell et al. 1975; Triska and Sedell 1975). This research is quite valuable for understanding processes and functions of LWD in northwestern and eastern streams. However, because of the varied climate and hydrologic regimes in the Southwest, research is needed on LWD dynamics and processes in the Southwest.

The Dude Fire, the largest and costliest fire in Arizona history, occurred below the Mogollon Rim on the Tonto National Forest as a result of a lightning strike on June 25th, 1990. The fire burned rapidly and ultimately consumed over 28,000 acres of forest type vegetation, destroyed over 50 homes, claimed six lives, and cost several million dollars of damage. The watersheds of three major permanent-flowing streams (Dude, Bonita, and Ellison creeks) were burned. The fire burned at a high intensity over 60 percent of the landscape, with the remaining 40 percent divided equally between medium and low intensity (personal comm., Grant Loomis, Tonto National Forest). Because of the severity of the burn, a large percentage of the undercover and overstory vegetation was destroyed and the soil became hydrophobic.

After the fire large amounts of dead trees were extant in riparian areas and land managers from the Tonto National Forest were concerned that debris accumulations would form. These, in turn, could wash out bridges and culvert crossings, and result in flooding of summer homes on Bonita and Ellison creeks. To mitigate the possible damage, larger trees were cut into smaller pieces on several creeks to facilitate flushing the pieces through the system more quickly and avoid debris jam formations. The question arises, "What is the best management practice with large woody organic debris in an area that

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has been impacted by a wildfire?"

Because of the lack of information and the question of debris jam formation, we began a study with two primary objectives: (a) to examine the relative transportation of differing sizes of hardwood and conifer LWD in four Southwestern watersheds—three burned and one unburned—and (b) to delineate the accumulation, composition, and movement of LWD in existing debris jams. This paper will only report on preliminary results of the first objective and future study methods for examining the second objective. Ultimately, data collected from this study will aid in the management of watersheds before and after wildfires in the Southwest.

Study Area

From July 6 to 10, 1990, runoff from initial, low-intensity monsoon rains, mixed with 5–10 cm of ash on the hillslopes and riparian areas, caused "slurry flows" down the study stream channels. As the monsoon intensity increased, stream discharge was in a flood state on July 10/11th and continued intermittently for 2 weeks. Prefire base flows in the study creeks averaged 0.1 m^3/sec during the July–September monsoon season, compared with base flows of approximately 10.6 m^3/sec during the early July flows in 1990 after the fire (Rinne in press).

In 1990 and 1991, water discharge was relatively high during both winter and summer months. During February–March 1993, 150–200-year flood events occurred on all drainages affected by the fire and others, such as Pine Creek 15 km to the west. These events eroded streambanks and transported both large (>1 m in length, >10 cm in width) and small (<1 m in length) woody organic debris. Since the large flows in 1993, precipitation and runoff have been relatively low and LWD pieces and accumulations have been basically static, except for movement of tagged LWD pieces on Dude Creek during flows in late August 1994 and in February 1995.

Presently, dead and decomposing trees subject to wind and weather continue to break and fall within the three burned drainages. In addition, trees have been uprooted because of a combination of loss of root systems, winds, and streamflow. In the process of breaking, falling, and uprooting, increasing amounts of LWD and sediment are entering the active stream channel.

Four sites were selected for the study: one study reach (1,500 m) each on Pine, Dude, Bonita, and Ellison creeks. Before the fire, the principal vegetation types in the overstory were ponderosa pine, juniper, and oak. The shrub overstory included three species of Arizona chaparral. The riparian vegetation was predominantly Arizona alder, Arizona walnut, big-toothed maple, and Arizona sycamore. To date, riparian vegetation, as well as ground cover within the drainage basins, is partially re-established.

Methods and Materials

From autumn 1993 through summer 1994, field work and data collection consisted of labeling, mapping, and description of LWD along each creek. Seventy-seven pieces were located and marked on Pine, 127 on Bonita, 114 pieces on Dude, and 150 pieces on Ellison. The number of pieces marked depended on both the availability of LWD and the goal of marking 150 pieces per creek. We selected LWD pieces of varying length and width marked within study sections on all creeks. By definition, a piece of woody debris had to be at least 10 cm in width at one end, and at least 1 m in length (Platts et al. 1987). The volume of pieces can ultimately be calculated by using the formula for the volume of a cylinder:

$$V = \frac{\pi \cdot (\text{diameter } 1 + \text{diameter } 2) \cdot \text{length}}{8}$$

The marking procedure was from downstream to upstream within the 1500 m linear reaches on each stream channel. A 60 cm wooden stake was set on the banks adjacent to the stream channels every 50 m and a 152 cm fence post at every 100 m. To indicate cumulative distance, fence posts were enumerated with a 5 cm circular, stamped aluminum tag; wooden stakes were labeled with a permanent marker.

Initially, to assess movement of LWD, each piece of wood was labeled with a 5 cm tag similar to those affixed to fence posts. Each piece of LWD has a unique, sequential number within that stream. Determining the initial location was necessary for calculation of downstream movement. Accordingly, a linear distance (made by laser or tape measurement), as well as a compass reading, from the metal tag to a labeled wooden stake or fence post was used to fix the initial location of each piece. Ultimately, all tagged pieces will be located in reference to the distance from the nearest stake on the bank, along the thalweg of the wetted channel. Visual observation and a metal detector are used to locate the

LWD pieces that have moved.

Each piece of LWD was classified as either hardwood or conifer. In this study, the relative distance traveled and the longevity of a hardwood versus a conifer LWD piece will be examined and delimited.

Because this study is programmed to last for 10 years, pieces of LWD that will not conceivably decompose or reduce in size more than 50 percent were chosen. The relative ages of the pieces were determined by (a) the percentage of bark present, (b) the degree of bark attachment on the LWD, and (c) the presence or absence of branches remaining.

Platts et al. (1987) was used to describe the position of each piece of LWD relative to the streambank: (a) complete bridge, (b) collapsed bridge, (c) ramp, and (d) drift (Figure 1). A complete bridge includes pieces that are suspended over the channel from one bank to the other at bankfull discharge. A collapsed bridge includes pieces that were once a complete bridge, but have broken and collapsed into the stream channel. Pieces of LWD with one side leaning on the bank and one side extended into the stream channel are included in the ramp category. Finally, the drift category comprises pieces that are lying within the channel.

The orientation of the most right side of the LWD piece relative to the right bank was spray painted or marked with a small metal tag. The positions of the LWD pieces relative to the stream channel were also recorded. Facing upstream, right bank, mid-channel, and left bank designations were assigned to pieces by dividing the stream bankfull width into thirds.

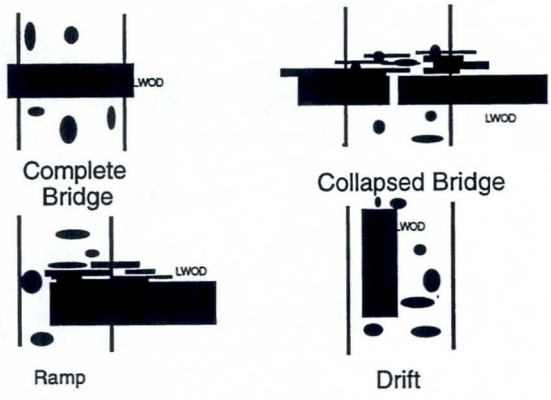

Figure 1. The Movement of LWD in Dude Creek in August 1994.

The last factor examined was the effect of the large woody debris pieces on the stream morphology. We determined whether a pool was created due to the presence of the piece, if sediment was being stored behind or in front of the piece, if the flow was being deflected by the piece or if it could be in higher flows, if other small woody debris or large woody debris was being stored behind the piece, and if there was no effect from the woody debris piece. A diagram was drawn for each piece of LWD. Diagrams indicate the direction of flow, the location of the piece in relation to the wetted channel, the bankfull discharge, bends in the channel, other pieces of LWD, debris jams, trees, and rocks.

In November of 1994, pressure transmitters coupled to data loggers were installed on each of the study creeks. Cross sections were surveyed and flow rating curves are being created.

Results

In August of 1994, flows approximating 90 m^3/sec occurred in Dude Creek and transported some LWD pieces a distance of 1500 m. No movement was detected on the other creeks. As a result of August flows on Dude Creek, 27 LWD pieces moved, while 8 pieces only shifted in position. Most pieces (15) moved less than 100 m; 8 pieces shifted in position only. Seven pieces moved 100–500 m, 3 pieces moved from 500–1000 m, and 2 moved greater than 1000 m.

A two-sample analysis (t-test) indicated a significant difference in the distance moved between pieces in the 1–2 m length class versus those in the greater than 6 m length class (Table 1). However, comparisons of movement between the 1–2 m length class versus the 4–5.9 m length class, and the 2–3.9 m length class versus the greater than 6 m length class, were not significantly different.

From the February 1995 flows on the study creeks, we have neither completed checking the streams for LWD movement nor calculated peak storm discharge measurements. On Pine Creek, we have examined 16 pieces for movement. Of these, 38 percent of the pieces could not be located, 56 percent did not move, and only 6 percent of the pieces did move (Table 2).

On Ellison Creek, we examined 101 pieces: 26 percent could not be located, 34 percent did not move, and 40 percent of the located pieces moved. On Dude Creek, out of the 74 pieces

examined, 11 percent were not located, 77 percent of the pieces did not move, and 12 percent of the located pieces moved. Bonita Creek has not been checked yet for movement.

Table 1. Analysis of the movement of LWD by size class on Dude Creek, summer 1994.

Comparison of Size Classes			Significance	
1–2 meters (4)	vs.	>6 meters (9)	0.02	Significant
1–2 meters (4)	vs.	4–5.9 meters (5)	0.16	Not Significant
2–3.9 meters (17)	vs.	> 6 meters (9)	0.08	Not Significant

Table 2. The movement of LWD as a result of winter (Feb. 1995) flows on two burned and one unburned watershed.

Streams Surveyed	Pieces Moved/ Not Located	Pieces Did Not Move	Pieces Moved/ Located
Pine (16)	38%	56%	6%
Ellison (101)	26%	34%	40%
Dude (74)	11%	77%	12%

Discussion

Based on the 35 pieces that moved on Dude Creek during the August 1994 flows, the two-sample analysis suggests that the length of LWD pieces is not the only factor influencing distance transported. Other factors that influence the transportation of LWD include the size of the piece, and the configuration and position in relation to the stream channel, flow, and other pieces and obstructions. The flows necessary to move a piece, channel configuration and morphology, and the instream obstructions (e.g. boulders, other LWD, standing trees, channel banks, bend in the channel, culverts) that may impede transport also affect mobility of LWD. Because of the complexity of factors affecting the transportation of singular LWD pieces and the need to understand debris jam formation and dissolution processes and dynamics, we will focus on the remainder of these components. Nevertheless, we will continue to monitor the movement of the large woody debris pieces already marked, their orientation before and after movement, and what obstructions have effected their movement.

Future Data Collection

To examine and monitor the movement and accumulation of LWD on burned and unburned channels, we need to first understand the variation in the quantity of LWD in undisturbed riparian stream systems in the Southwest and the processes involved in the transportation and accumulation. Next we need to examine burned channels to see the differences between each channel before comparing them together and then to the unburned system. To gain a baseline understanding of the location, size, quantity, movement, and debris jam formations of LWD pieces in southwestern riparian areas we will survey additional creeks within the region below the Mogollon Rim to define the natural variability of LWD in these stream channels.

Ultimately the question is, "What is the best management practice with the large accumulation of LWD after a wildfire?" This study is planned to continue for 10 years. Results of this study will provide an understanding of the transport and accumulation of LWD in the Southwest, as well as the effects of fire on these processes. Conceivably, this knowledge will enable managers to develop the best management practices in and along aquatic-riparian ecosystems before and after fires.

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SEDIMENT AND NUTRIENT REGIME FROM A CENTRAL ARIZONA CHAPARRAL WATERSHED

Steven Overby and Malchus B. Baker Jr. [1]

A critical problem in the Southwest is determining the effects of increased management activities on the delivery of sediment, water, and nutrients from side slopes into stream channels. Prescribed fire is one of the tools used in the conversion of chaparral to grass-shrub communities for the purpose of increasing water yields, increasing forage for domestic animals and wildlife, enhancing wildlife habitat, and reducing fuel loads (Brown et al. 1974; Hibbert and Davis 1986). Prescribed fires are important in achieving the above objectives, but not all effects are positive. There is a need to better understand the interaction of fire with hydrologic processes that affect our watersheds and the resources they provide (Tiedemann 1978).

Erosion is a serious problem because of detrimental effects to soils but also because of harmful effects on water quality. Soil erosion reduces site productivity by removing nutrient-rich surface horizons, and changes available plant water that is so critical in arid regions. Plant cover is a major factor influencing overland flow. When landscapes are denuded either naturally or due to man's activities, increased overland flow strips these surface horizons, and can eventually lead to gully incision or mass wasting on steep slopes. These surface horizons are critical to the long-term site productivity of an area because of their influence on available water and nutrients needed by plants. As overland flow removes these nutrients and organic matter, soil water-holding capacity is reduced, and plant establishment becomes increasingly more difficult, which further destabilizes the affected landscape.

Movement of sediments from chaparral slopes to upstream tributary channels and eventually out of the watershed is a complex process that is primarily dependent on amounts and intensities of precipitation (Heede 1988). Soil particles are initially transported from slopes to upstream tributaries through the processes of downslope soil creep, dry ravel, and overland flow (Hibbert et al. 1974). Sediments accumulate in the upstream tributaries until streamflow of sufficient magnitude occurs to transport the sediments to downstream channels and out of the area.

Sediment production and transport from chaparral watersheds in Arizona is produced during periods of heavy precipitation, which usually occur in winter and early spring as a series of cyclonic storms, often occurring one after another in rapid succession (Hook and Hibbert 1979; Heede 1988). Precipitation from these storms eventually exceeds the water-holding capacity of the soil, thereby producing flow rates capable of transporting sediment downstream. Only rarely are summer storms big enough to generate and sustain the quantity of overland flow required to transport large amounts of sediment any significant distance downstream. These smaller flow events gradually move sediment downslope, then normally dissipate, dropping their sediment loads in the flatter, dry sections of the channel system. Streamflow of sufficient magnitude and duration to transport sediments to downstream channels and out of the area is more likely to be produced by a series of intense winter storms that result from above-normal precipitation. An exception is immediately after a chaparral wildfire, when overland flow occurs much more readily as a result of a fire-induced hydrophobic layer near the soil surface (DeBano 1981; Scholl 1975).

The objectives in this study were (a) to determine the rates of sediment loss from chaparral-covered watersheds, and (b) to identify the factors influencing these rates, both natural and induced by man.

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Research Area

The study area lies within the Battle Flat watershed in the Bradshaw Mountains on the Prescott National Forest in central Arizona. The Prescott National Forest and the Rocky Mountain Forest–Range Experiment Station implemented a pilot demonstration project in this area for testing and refinement of technology to manage chaparral watersheds as grassland–brushland mosaics for increased multiresource outputs in Arizona (Krebill and Tackle 1978). The Battle Flat watershed, at a median elevation of 1,633 m, comprises several contiguous subwatersheds (Figure 1). Water yield is generally intermittent with streamflow beginning in early winter during wet years and continuing through May or June. In dry years little or no flow occurs. Mean annual precipitation in the Battle Flat Watershed is 631 mm (1980–1987). Precipitation generally occurs as cyclonic storms in winter and local convective storms in the summer.

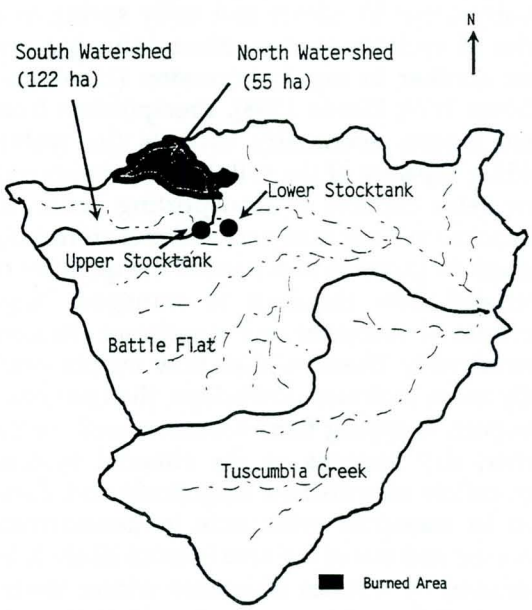

Figure 1. Battle Flat Watershed.

The two ungaged subwatersheds used in this study occupy 177 ha in the northwest portion of Battle Flat watershed and the upper 67 percent of a gaged subwatershed (262 ha) (Figure 1). The south subwatershed (122 ha) is over twice the size of the north subwatershed (55 ha). The general geologic composition of these subwatersheds is described by Anderson and Blacet (1972) as massive bedded crystalline tuff with recent gravels along stream channels. The aspect is generally southeast, and slopes range 15–40 percent with some as high as 60 percent. Elevations range from 1,586 to 1,769 m above sea level.

Soils on the two subwatersheds, which are similar, include Moano gravelly loam and Moano-Lynx association in the areas of lower elevation and Moano very rocky loam on the upper slopes. These soils are classified as loamy-skeletal, mixed, nonacidic, mesic Lithic Ustorthent (Humbert et al. 1981) that developed from a mixture of Spud Mountain Volcanics of the Yavapai series of Precambrian rock and alluvium (Anderson and Blacet 1972).

Developing a fire history for a particular chaparral stand is difficult because fire-scarred material is not available that can accurately date fires using standard dendrochronology techniques. The Battle Flat area presented a unique opportunity in that a reliable record exists of historical fires burning in a ponderosa pine (*Pinus ponderosa*) stand surrounded by chaparral (Dietrich and Hibbert 1988). Fire history information from this ponderosa pine stand affords indirect evidence of the role of fire in chaparral stand development. Dietrich and Hibbert (1988) examined the fire history of the Battle Flat area by separating premining, mining, and postmining periods.

Dietrich and Hibbert (1988) found an approximate 2-year fire interval in this ponderosa pine stand during the premining period (prior to 1863). This fire frequency interval created a fairly open stand with a grass-forb ground cover. This ground cover was the fuel source needed to spread the fire after ignition by lightning. As these fires reached the edge of the pine stand, chaparral stands with adequate fuel loads to continue the fire would burn. These chaparral stands are speculated to have been more than 10 to 15 years old when the amount of dead material is adequate to sustain a fire.

Placer mining in the Bradshaw Mountains (1863–1885) heavily utilized the pine–oak–juniper stands for mine timbers, construction lumber, and fuel. Dietrich and Hibbert (1988) were unable to find enough fire-scarred material to piece together an adequate fire history for this time period. No living trees older than 100 years were found, further supporting the conclusion of heavy utilization for mining.

Little mining continued after 1885, but grazing and frequent fires during the late 1800s maintained a natural mosaic that aided early efforts of fire suppression. By the 1920s the natural mosaic disappeared as chaparral stands became overmature, developing into large continuous stands greater than 20 years old. The Battle Flat watershed is in this overmature state with a large amount of fuel maintained by aggressive fire suppression.

Prior to burning, plant cover on the study area was mainly shrubs with minor amounts of understory forbs and grasses. Prefire shrub cover was primarily shrub live oak (*Quercus turbinella*; 48%), birchleaf mountain mahogany (*Cercocarpus betuloides*; 27%), and pointleaf manzanita (*Arctostaphylos pungens*; 19%). The remaining 6 percent shrub cover was made up of several shrub species. Dimensional analysis showed that the mean oven-dry weight of total, annual twigs, and leaves for mountain mahogany was 10,062, 259, and 1,154 kg/ha, respectively (Whysong and Carr 1987). An unknown quantity of pointleaf manzanita and other shrubs was also present.

Annual precipitation measured at a permanent weather station, located on the north subwatershed, averaged 686 mm over a 10-year period preceding the prescribed fire. Arizona chaparral is characterized by two distinct wet periods with about 55 percent of the moisture occurring in the cool winter months (November to April) as frontal storm events and 35 percent in July through September in the form of local convection storms. October, May, and June are the driest months, receiving only about 10 percent of the annual precipitation (Hibbert et al. 1974). Temperatures recorded at the weather station ranged from as low as -12°C to a high of 19°C in the winter, while summer temperatures ranged from a low of 2°C to a high of 33°C.

The gaged subwatershed containing the two ungaged study watersheds yielded 28 mm of water annually between 1980 to 1988, which was 5 percent of the average precipitation (Table 1). In wet years much larger amounts are yielded (over 80 mm in 1980 and 1983), and in dry years very little (7 mm or less during 3 of the 8 years of record). Water enters the ephemeral channel system as both subsurface and surface flow. In years of above-normal precipitation, subsurface flow predominates over surface runoff, and streamflow may extend into the summer. Even after surface flow in the channels ceases, subsurface drainage often continues for a time through deep channel alluvium that can exceed 3 m in depth. Loss of ground water to deep seepage is generally prevented by low permeability bedrock.

Table 1. Precipitation and runoff on study area.

Water Year	Precipitation (mm)	Streamflow (mm)
1980	791	86
1981	342	7
1982	807	13
1983	873	90
1984	458	10
1985	729	12
1986	566	3
1987	482	7

Two stock watering tanks are located at the mouth of the ungaged subwatersheds (Figure 1). These small reservoirs are usually dry by mid summer. The lower stock tank was constructed in 1964 and, until 1977, received all sediment yields from both subwatersheds. Another stock tank (upper) was constructed in 1977 and is located about 210 m upstream from the lower tank. The upper tank now receives sediment from the south watershed and the lower tank receives sediment from the north watershed.

The 55 ha north subwatershed was prescribed burned over a 2-day period on October 30–31, 1985 using a helitorch. Relative humidity during burning ranged from 10 to 30 percent and air temperatures were between 16 and 29°C. Wind speed was 8–29 km/hr. The fuel moisture content of new growth prior to burning was 100 percent and 72 percent and for old growth was 82 percent and 61 percent in shrub live oak and mountain mahogany, respectively. The combination of fuel and meteorological conditions produced marginal burning on north-facing slopes, although south-facing slopes burned intensely. During the most intense fire activity, the rate of spread was about 12 m/min with flame heights in excess of 4 m.

Methods

The volume of sediment deposited in the stock tanks was determined from surface area measurements and depth samples (Hook and Hibbert 1979). The dates of construction of the two stock tanks and the total volume of sediment deposi-

tion provide the information needed to determine baseline sediment yields from the study watersheds (1964 through 1978). In July 1985 the first of four surveys was made of the nutrients contained in various layers of sediment deposited in the tanks. Additional surveys of surface deposits were made in November, 1986, one year after burning, and again in April 1988.

In July 1985, both upper and lower stock tanks were sampled by trenching with a small backhoe along permanent survey transect lines. Samples were taken from trench walls at 0–13 cm and 13–25 cm in both stock tanks. Only transects with large enough accumulations were trenched. Additional samples below 25 cm were taken where possible to provide more information. The subsequent samples of surface material were collected when basin surveys done prior to the sampling showed new accumulation of sediments.

Bulk density samples were taken by driving metal rings vertically into the trenched wall. These samples were placed in paper bags, oven dried at 105°C, and weighed. Additional samples were taken from the same layer as the bulk density samples to be analyzed later for total nitrogen, phosphorus, calcium, potassium, and magnesium. These samples were analyzed by a commercial laboratory. Total nitrogen was determined by the Kjeldahl method. Total phosphorus, calcium, magnesium, and potassium were determined by atomic absorption on a nitric-perchloric acid wet ash sample.

Results and Discussion

Hook and Hibbert (1979) determined an annual rate of sediment production for these subwatersheds of 0.7 m^3 ha^{-1} over a period of 14 years, realizing that 14 years was too short a period to establish a reliable long-term erosion rate when yearly sediment production is known to vary from zero to at least 2.9 m^3 ha^{-1}. To produce the maximum rate on these subwatersheds, approximately 444 mm of precipitation fell, which saturated the soil mantle with an additional 261 mm to produce runoff. Most of the 2.9 m^3 ha^{-1} of sediment was deposited in March 1978 after over 850 mm of precipitation occurred from the middle of December to the end of March. Hook and Hibbert (1979) identified four winter seasons, during the period 1964 to 1978 (prior to instrumentation of Battle Flat), when similar precipitation amounts were observed at Crown King (located 7 miles south of Battle Flat). Since Battle Flat has been instrumented beginning in 1979, two additional seasons, winters of 1979–80 and 1982–83 (Table 1), had similar seasonal precipitation, showing the highest sediment production prior to the 1985 prescription fire (Table 2).

The stock tank receiving sediment from the burned watershed attained its highest production after the 1987–1988 winter, 2 years after burning (Table 2). No net accumulation was recorded the first year after burning. The corresponding precipitation (Table 1) demonstrates the high variability and unpredictable nature of sediment production as 1985 following the fire had minimal accumulation and 1987 had no net accumulation yet higher total winter precipitation. The highest sediment yields occurred during the summer rains of 1986, a result of a hydrophobic soil layer due to fire. High sediment yields were expected following the fire, but little accumulation the year following and no net accumulation after the 1986–1987 winter precipitation period demonstrates how extremely variable these systems are. Hibbert (1985) reported a 25-fold increase in sediment yield following a wildfire in Arizona chaparral. The Battle Flat subwatershed yielded 8.28 kg $ha^{-1}yr^{-1}$ for the 3 years following the prescribed fire, compared to 0.73 kg $ha^{-1}yr^{-1}$ for the 7 years prior.

The chemical concentrations of sediments can be quite different than the soil on the watershed slopes. As colluvial material that has been deposited in the upstream channels is moved downstream, the finer fraction and more soluble constituents of the sediments stay suspended longer before being redeposited further downstream. Our sampling focused on the areas in the stock tanks with the greatest deposition. These areas are located close to the dam where the greatest depth of water also occurs. For this reason, the chemical composition of the sampled sediments was higher in cation concentrations as this finer soil fraction has a higher cation exchange capacity, solubility, and movement, with an interflow of nutrients redeposited when the impounded water in the stock tanks evaporates.

A significant increase for all of the measured nutrient concentrations (Table 3) except magnesium and potassium following the fire is related to the destruction of organic material. The post-fire increases in total nitrogen, phosphorus, and calcium are due to the mineralization of shrubs and litter during the fire. These mineralized

Table 2. Sediment Accumulation at Battle Flat Stock Tanks.

		Accumulated Total (m^3)		Total (kg $ha^{-1}yr^{-1}$)	
Survey	Date	Lower	Upper	Lower	Upper
BUILT		1964	1977		
1	09/78–09/79	1690	349	0.86	3.34
2	07/80–11/80	210	289	2.85	2.77
3	11/81	18	132	0.45	1.26
4	08/83	69	62	0.97	0.34
5	04/85–01/85	*	88	*	0.66
TRENCH	07/85				
BURN	11/85				
6	05/86	2	*	0.05	*
7	10/86	236	*	11.60	*
8	06/87	*	*	*	*
9	03/88	542	32	13.30	0.30
10	05/89	366	71	8.17	0.63

* No sediment accumulation

Table 3. Average Total Nutrient Concentrations (%) in Lower Battle Flat Stock Tank.

		Year				
Nutrient	n	64–78*	78–81*	81–85*	85–86	86–88
Nitrogen	35	0.048a	0.044a	0.063b	0.280c	0.247c
Phosphorus	35	0.027a	0.025a	0.029a	0.032b	0.032b
Calcium	35	0.366a	0.365a	0.411b	1.005d	0.590c
Magnesium	35	1.079a	1.121a	1.178a	1.029a	1.077a
Potassium	35	0.653a	0.650a	0.722a	0.779a	0.778a

*Prefire samples
Means with different letters significantly different ($p < 0.05$) as determined by Tukey's mean separation test.

Table 4. Prefire Nutrient Totals (Kg) in Battle Flat Soils and Sediments.

	Nitrogen	Phosphorus	Calcium	Magnesium	Potassium
Soil (0–10 cm)	1361	256	1829	374	249
64–78	10.6	5.9	80.4	237.1	143.5
78–81	1.3	0.7	10.8	33.2	19.3
81–85	0.6	0.3	3.7	10.6	6.5
85–86	8.7	1.0	31.1	31.8	24.1
86–88	17.4	2.3	41.6	75.9	54.8

nutrients are deposited as ash on the soil surface and are easily transported into the channel by overland flow.

When compared with a prescription fire in California chaparral (DeBano and Conrad 1976), percentages lost of total nitrogen, phosphorus, magnesium, and calcium in the soil and ash layer were lower, while potassium was the same for the Battle Flat area. DeBano and Conrad (1976) reported losing 3.4% nitrogen, 2.4% phosphorus, 9.7% potassium, 3.5% calcium, and 10.9% magnesium in the first year while the Battle Flat subwatershed showed a 0.6% nitrogen, 0.4% phosphorus, 9.7% potassium, 1.7% calcium, and 8.5% magnesium loss following the fire. During the next sample period, ending 3 years after the fire, percentages lost were 1.3% nitrogen, 0.9% phosphorus, 22% potassium, 2.3% calcium, and 20.3% magnesium for Battle Flat. The California losses decreased with time while the Battle Flat losses increased. Runoff and sediment yields for Battle Flat were much lower than the California site but this is directly related to lower precipitation (72.9 mm versus 390.9 mm). DeBano and Conrad (1978) reported nutrient loss differences with varying fire intensities. This increase in nutrient losses occurs because wildfires typically burn at much higher temperatures than prescribed fires, mineralizing more litter and biomass, which is easily transported off the watershed.

Conclusions

The use of fire as a tool for managing chaparral ecosystems can improve wildlife habitat by stratifying the canopy and providing openings for herbaceous plants and grasses. The reduction of fuel loads that could lead to catastrophic wildfires is also a positive benefit of prescription fire. An increase in sediment production following prescription fire is significant for a small burned subwatershed, but the total increase for the watershed as a whole is small. If sediment yield as found by Hibbert (1985) is applied to large chaparral wildfires such as the Battle Fire in 1972, two and a half times the amount of sediment and associated nutrients would be removed from chaparral watersheds as compared to prescribed fire. Higher nutrient losses would adversely impact not only downstream water quality, but also vegetative recovery and sustainability of these watersheds. These negative impacts can be extended or ameliorated over a much greater time period by the use of prescribed fire, which more closely approximates the natural fire regimes of chaparral ecosystems.

The information currently available on fire effects indicates that it is desirable to burn Arizona chaparral when conditions protect the litter layer and soil from extreme temperatures and extended durations of heating (DeBano 1988). Whenever possible, prescribed fires in Arizona chaparral should be planned during conditions of cool air temperature, high humidity, high fuel moisture, and low wind speed. These marginal burning conditions may be hard to achieve but are desirable to mitigate the negative impacts of prescribed burning.

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Environmental Fate and the Effects of Herbicides in Forest, Chaparral, and Range Ecosystems of the Southwest

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Biological methods, fire, herbicides, and mechanical methods have all been studied in an effort to determine appropriate ways of manipulating arid land vegetation for improvement of wildlife habitat, streamflow and water yield, increasing forage for livestock, and enhancing recreational benefits and scenic diversity. Because water is ultimately essential for all of these uses, and because of an increasing concern over the availability of water for human consumption, augmentation of water yield has been a significant part of many of these studies. Many of the studies conducted between the mid-1950s and mid-1970s have been reviewed (Hibbert et al. 1974; Hibbert 1979). Other research on water yield augmentation in the West has been reviewed by Anderson et al. (1976), Harr (1983), and Kattelmann et al. (1983). Water yield can be increased by managing snow and by managing vegetation. Managing snow requires reducing the evaporative loss from snow, and/or redistributing snow pack (by fencing, etc.) to increase the snow melt and runoff to streams. Deep-rooted woody perennial species, notably shrub live oak (*Quercus turbinella* Greene), birchleaf mountain mahogany (*Cercocarpus betuloides* Nutt.), sumac (*Rhus ovata* S. Wats.), hollyleaf buckthorn (*Rhamnus crocea* Nutt.), Emory oak (*Q. emoryi* Torr.), yellowleaf silktassel (*Garrya flavescens* S. Wats.), New Mexico locust (*Robinia neomexicana*) and similar species tap soil water and through evapotranspiration greatly reduce potential water yield. The impact of vegetation on evapotranspiration losses can be reduced in the most drastic terms by conversion of the cover type to one that demands less water, or by reducing the vegetation density. However, even type conversion does not always give significant or economically useful increases in water yield. The areas with the greatest potential for water yield increases due to vegetation manipulation are those with greater than 450 mm (>18 in) of annual precipitation and where potential evapotranspiration exceeds 380 mm (15 in) per year (Hibbert 1979). Thus, in general, the warmer and moister locations provide the greatest opportunities for increased water yield.

Herbicides have been used in the Southwest for more than 50 years to manipulate forest, range, and chaparral vegetation. The purpose of this paper is to review that use, identify compounds that have been used historically, and discuss the impacts of those still in use on water quality.*

Herbicides for Vegetation Management in the Southwest

Buthidazole, fenuron, karbutilate, and 2,4,5-trichlorophenoxy acetic acid were tested and in some cases put into use for controlling chaparral species. However, they are no longer available for use. They are covered here to give an historical perspective on the use rates for the more commonly used chemicals. Buthidazole was tested for efficacy against fire sprouts of shrub live oak. It was found to be effective at rates of 4.5 and 9.0 kg/ha active ingredient (ai) (4 and 8 lb ai/ac; Davis et al. 1980). This compound was generally not tested against more mature forms of live oak.

Fenuron, a substituted phenylurea herbicide, was found to be somewhat effective against shrub live oak, and mountain mahogany when applied in the winter at 17.9 kg/ha ai (16 lb ai/ac; Davis and Pase 1969). Spot treatment

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* Although this report discusses research involving pesticides, such research does not imply that the pesticide has been registered or recommended for the use studied. Registration is necessary before any pesticide can be recommended. If not handled or applied properly, pesticides can be injurious to humans, domestic animals, desirable plants, fish, and wildlife. Always read and follow the directions on the pesticide container.

enabled selective control of undesirable species, but eradication of the target species was not possible, even at this high rate. Hibbert et al. (1982) reported excellent control of shrub live oak and mountain mahogany with 25.8 kg/ha ai (23 lb ai/ac) in the conversion of chaparral to grass cover. Application was by hand using 25 percent ai pelleted fenuron distributed beneath the shrubs. Follow-up treatment was not necessary.

Karbutilate, a carbamate herbicide, was used in the control of woody species at rates up to 22.4 kg/ha ai (20 lb ai/ac). Davis (1982) reported that control of shrub live oak and mountain mahogany was excellent 3 years after treatment. Davis et al. (1980) documented that control of mature shrub live oak was also accomplished with 4.5–9.0 kg/ha ai (4–8 lb ai/ac) when applied in the summer, but control of shrub live oak fire sprouts was best when application was made in mid-winter.

The phenoxy herbicide, 2,4,5-T, was the most-used herbicide in the history of brushy species control in the U.S. Because of environmental concerns it is no longer registered by the United States Environmental Protection Agency for use in the U.S. Applied at rates of up to 13.4 kg acid equivalent (ae)/ha (12 lb ae/ac) for general weed control, it was frequently used in multiple applications of 2.2 kg ae/ha (2 lb ae/ac) for control of many brushy species (Elwell 1964). DeBano et al. (1984) report that four annual applications of 1.7 kg ae/ha (1.5 lb ae/ac) for the control of shrub live oak fire sprouts and birch-leaf mountain mahogany (*Cercocarpus betuloides*) resulted in only 42 percent and 72 percent control, respectively. Ingebo and Hibbert (1974) report the use of a 6.6 percent solution of 2,4,5-T as a basal spray on desert ceanothus and shrub live oak. Spot treatment was required in the second year following initial treatment for good control. While it is difficult to relate a solution composition to an area rate, use of 187 L/ha (20 gallons/ac) of such a solution would equal approximately 11.8 kg ae/ha (10.5 lb ae/ac). Control of chamise (*Adenostoma fasiculatum*) was accomplished with 4.5 kg ae/ha (4 lb ae/ac) (Plumb et al. 1977)

Picloram and 2,4-D are two phenoxy acetic acid herbicides that were commonly used for brush control. Picloram is a chlorinated picolinic acid, most frequently used in chaparral vegetation control as the 10 percent ai pelleted formulation of the potassium salt. Picloram pellets applied at the rate of 10.4 kg ai/ha (9.3 lb ai/ac) gave marginal control of shrub live oak, but were more than was needed for other chaparral species (Davis et al. 1968). Picloram was also frequently used in combination with 2,4-D. Picloram (2.8 kg ai/ha or 2.5 lb ai/ac) and 2,4-D (5.6 kg ae/ha or 5 lb ae/ac) were applied to pinyon–juniper woodland by backpack mist blower. This treatment gave good control of juniper, but shrub live oak and pinyon recovered after 2–3 years (Baker 1984).

Tebuthiuron, a substituted urea herbicide, is still used at rates up to 4.5 kg ai/ha (4 lb ai/ac) for control of most woody species, and at rates up to 17.9 kg ai/ha (16 lb ai/ac) for control of hard-to-kill perennial weeds. In general, higher rates are required for weed control on deep fine-textured soils than on shallow coarse soils (Herbel et al. 1985). Many species of the Southwest can be controlled with 0.6–1.1 kg ai/ha (0.5–1.0 lb ai/ac), including creosote bush (*Larrea tridentata*), whitethorn acacia (*Acacia constricta*), tarbush (*Flourensia cernua*), skunkbush (*Rhus trilobata*), and ocotillo (Herbel et al. 1985). In a comparison study on juniper and pinyon, Johnsen and Dalen (1990) found that tebuthiuron treatment gave better control than picloram at the same rates, but tebuthiuron was also more damaging to understory plants. When compared with picloram for control of sand shinnery oak (*Q. havardii*), tebuthiuron gave acceptable control at rates of 0.3–1.1 kg ai/ha (0.3–1.0 lb ai/ac), while rates of 2.2 kg ai/ha (2 lb ai/ac) of picloram were required to give similar control (Jacoby and Meadors 1982). Similarly, Jones and Petit (1984) found that 0.5 kg ai/ha (0.4 lb ai/ac) gave 98 percent control of sand shinnery oak, and resulted in increased grass production in the second year after treatment. Davis et al. (1980) determined that shrub live oak requires up to 9 kg ai/ha (8 lb ai/ac) depending on the level of control required and whether the target vegetation is mature brush (higher rates) or fire sprouts (lower rates). In a brush-to-grass conversion study, Davis (1982) used 4.6 kg ai/ha (4.1 lb ai/ac) tebuthiuron to control shrubs left over from an earlier (10 years) treatment with fenuron. Sagebrush (*Artemisia tridentata* and *A. cana*)

was controlled with 1.1 kg ai/ha (1 lb ai/ac) of tebuthiuron while most associated grasses were not damaged (Whitson and Alley 1984).

Water Yield

The main objective of herbicide use on forest lands in most parts of the country is control of competing vegetation during postharvest regeneration. When herbicides were used more extensively in the Southwest, the principal goal of herbicide use was water yield augmentation. Most of the studies of herbicide effects on water yield were done in chaparral ecosystems (Hibbert et al. 1974; Hibbert et al. 1975; Hibbert 1983; Davis 1984; Hibbert and Davis 1986; Ingebo 1972; Ingebo and Hibbert 1974). Clary et al. (1974) and Baker (1984) reported on water yield responses to herbicide application in pinyon–juniper ecosystems.

The results of water yield research in chaparral using fire and herbicides were summarized by Hibbert et al. (1974, 1975) and Brown and Fogel (1987). Davis (1984) reported that application of 22 kg ai/ha of karbutilate to chaparral on the Three Bar watersheds increased runoff efficiency (percent of precipitation yielded to stormflow). The ratio doubled from 0.56 (pretreatment) to 1.3 (post-treatment). Over a 3-year period the mean annual water yield increase amounted to 703 percent. Hibbert and Davis (1986) documented a streamflow increase of 68 mm/yr (+72%) in the chaparral-covered Whitespar watersheds after application of 3.4 kg ai/ha tebuthiuron to 55 percent of one watershed. Application of a 6.6 percent solution of 2,4-D and 2,4,5-T on two chaparral watersheds at Sierra Ancha resulted in a 22–35 percent increase in streamflow (Ingebo and Hibbert 1974). For chaparral ecosystems in Arizona, water yield increases from herbicide treatments (up to 703% increase) were equivalent to those from fires (up to 700% increase), and were dependent on watershed aspect, annual precipitation, vegetative cover reduction, and season. Most streamflow increases came during the winter months, and some ephemeral watersheds produced perennial flow.

Clary et al. (1974) and Baker (1984) reported on streamflow changes to a Beaver Creek pinyon–juniper watershed after application of 2.8 kg ae/ha of picloram and 5.6 kg ae/ha 2,4-D. Compared to an untreated control, the herbicide-treated watershed produced 130 percent more storm runoff, 25 percent higher peakflow, and greater basin recharge during a 20 to 50-year storm (Baker 1984). The herbicide effect was a combination of greatly reduced evapotranspiration losses due to foliage leaf area reduction and shading by standing dead stems.

Water Quality: Herbicide Residues

Concentrations of fenuron (Davis and Ingebo 1970), karbutylate (Davis 1975), picloram (Davis and Ingebo 1973), and tebuthiuron (Emmerich et al. 1984) in streamflow and storm runoff in the Southwest have been studied at a number of locations (Table 1). Measured peak concentrations fall in the same low range (0–820 mg/m^3) as those reported for forestry herbicides elsewhere in North America. These peaks were low enough and of such short duration that they did not pose a water pollution problem. Persistence in runoff documented in the studies reported here ranged from 14 months (picloram) to 72+ months (tebuthiuron). The latter herbicide is particularly persistent—hence its utility as a brush control herbicide. Total losses of herbicide residues in streamflow ranged from less than 0.5 percent to as high as 4.5 percent. The losses at the upper end of the range exceed those measured in the South or Pacific Northwest (Neary and Michael 1993).

Sediment

The use of herbicides for weed control or species type conversions usually does not result in significant sediment yield increases (Neary and Hornbeck 1994). However, in the arid Southwest widely separated variations can occur. Natural sediment yield rates can range from 0.002 to 2.5 Mg/ha/yr. Ingebo and Hibbert (1974) reported an eight-fold reduction in sediment loss from a herbicide-treated chaparral area due to grass growth along channel margins. But, Hibbert at al. (1974) noted a ten-fold increase in sediment yield due to channel adjustments to higher flows and slope instability on steep upper slopes. By contrast, wildfires in the Southwest can increase sediment loss rates by factors of 1,000–369,000 times normal (Table 2).

Nitrates

The side effects of killing vegetation include interruption of major nutrient cycling pathways, increased organic matter decomposition, and increased microbial activity. These effects can

Table 1. Maximum herbicide concentrations in streamflow from operational applications to forests and woodlands.

Herbicide	Application Rate (kg/ha)	Concentration (mg/m^3)	Reference
Southwest USA			
1. Fenuron	27.8	430	Davis and Ingebo 1970
2. Karbutilate	2.2	51	Davis 1975
3. Picloram	9.3	370	Davis and Ingebo 1973
4. Tebuthiuron	0.8	91	Emmerich et al. 1984
6. 2,4-D	*	*	
5. 2,4,5-T	*	*	
Other Locations			
7. Glyphosate	1.3	270	Newton et al. 1984
8. Hexazinone	3.6	820	Legris 1988
9. Imazapyr	2.2	680	Michael and Neary 1993
10. Picloram	5.6	442	Michael et al. 1989
11. Sulfometuron Methyl	0.4	44	Michael and Neary 1993
12. Triclopyr	0.9	620	Wan 1987
13. 2,4-D	2.2	132	Norris 1967
14. 2,4,5-T	4.5	200	Norris 1981

* No information available.

Table 2. Sediment loss and herbicides.

Location	Treatment	First Year Sediment Yield	
		Control (Mg/ha)	Treated (Mg/ha)
Florida	Cut, Herbicide	0.022	0.010
North Carolina	Cut, Herbicide	0.044	0.165
Mississippi	Cut, Herbicide	0.134	0.133
Georgia	Herbicide, Cut	0.067	0.170
Arkansas	Cut, Herbicide	0.071	0.251
Arizona	Herbicide	0.019	0.002
Arizona	Herbicide	2.565	2.049
Arizona	Wildfire	2.200	50.500

impinge on water quality by releasing anions and cations into streamflow. This can best be illustrated by examining nitrate nitrogen (NO_3-N) concentrations in streamflow.

On the chaparral watershed at Three Bar treated with fenuron (20.5 kg ai/ha), karbutilate (7.5 kg ai/ha), and tebuthiuron (0.9. kg ai/ha), Davis and Debano (1986) found high soil NO_3-N levels (82.9 mg/L) to depths of 4.6 m. The mean annual streamflow NO_3-N concentration peaked at 11.7 mg/L, with an instantaneous NO_3-N maximum of 18.4 mg/L (Davis 1987). Elevated NO_3-N concentrations persisted for over 14 years. These high NO_3-N levels are among the highest reported for disturbances to forest ecosystems (Neary and Hornbeck 1994). The main environmental consequences of the prolonged release of NO_3-N from chaparral watersheds are downstream eutrophication and drinking water quality standard violations. No real problems developed as a result of these vegetation conversions due to downstream dilution from untreated areas and the limited extent of chaparral conversions.

Summary and Conclusions

Buthiazole, fenuron, and karbutilate were used in the early research on arid land vegetation control, but the rates required were quite high (up to 22. kg ai/ha–20 lb ai/ac). While they are no longer available for use, they serve to demonstrate the advances made over the last 20 years in herbicide technology. Extremely high rates like those reported for these compounds have been replaced with the much lower rates of newer chemistry. The phenoxy herbicides (2,4,5-T and 2,4-D) were used at intermediate rates (2.2–11.4 kg ae/ha or 2–10 lb ae/ac), but adequate vegetation control frequently required multiple applications. Picloram provided another advance in treatment rate reduction requiring rates of 2.2–5.6 kg ai/ha or 2–5 lb ae/ac, but also required some follow-up treatments. Tebuthiuron is active against many arid land species at rates as low as 0.28 kg ai/ha (0.25 lb ai/ac), and is useful at rates approximately half that required by picloram. Because of the long soil residue time of tebuthiuron, follow-up treatments are usually not needed.

Water yields in the Southwest were increased by 22 to 703 percent by the application of herbicides to mainly chaparral watersheds. The amounts were a function of watershed aspect and elevation, annual precipitation, the percent and amount of vegetation cover removed, and soil depth. Peak flows and flow duration (especially during winter) were increased by herbicide use. Residues of fenuron, karbutilate, and tebuthiuron were measured in streamflow for durations of 14 to 72+ months. The maximum concentrations ranged from 41 to 430 mg/m^3, depending on the application rate. Sediment yields from herbicide use in the Southwest have been increased and decreased, but are nowhere near the magnitude of increased yields produced by wildfires. Concentrations of NO_3-N in streamflow were considerably elevated and persisted for 14+ years.

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FOSSIL CREEK: RESTORING A UNIQUE ECOSYSTEM

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Relicensing of the Childs/Irving Hydroelectric Project (Childs/Irving) provided an opportunity for restoration of a unique travertine ecosystem in central Arizona. Childs/Irving, located on Fossil Creek in Gila and Yavapai counties, has generated electricity since the early part of this century. The original hydropower license for Childs/Irving was issued on January 1, 1945 by the Federal Power Commission for a 50-year period. In 1994, Arizona Public Service Company (APS) applied to the Federal Energy Regulatory Commission (FERC) to relicense the project. The relicensing and associated environmental assessment process provided a rare opportunity to restore a unique, but degraded travertine ecosystem by restoring streamflow to a mostly dewatered creek. This paper describes many of the unique features of the ecosystem and some of the issues that surfaced during the environmental assessment process.

The Project

From its origin in the incised canyons of the Mogollon Rim country just north of Strawberry, Arizona, Fossil Creek flows in a southwesterly direction for 17 miles before entering the Verde River below Childs. It flows on lands entirely under the jurisdiction of the National Forest Service, and forms the boundary between the Coconino and Tonto national forests for much of its length. Fossil Creek is an intermittent stream from its headwaters until it reaches Fossil Springs, which are located approximately one third of the way down the mainstream of the creek. The springs emerge over a 1,000 foot reach of the creek, at a constant discharge of 43 cubic feet per second (cfs) and a constant temperature of 72°F.

In the early 1900s the flow from these springs was diverted from the creek and used to generate hydroelectric power for the mines near Prescott, Jerome, and Crown King. Water is diverted at Irving Dam, 0.2 miles downstream from the springs, and is transported 10.4 miles by a series of open flumes, syphons and penstocks to power plants at Irving and Childs. The diverted water is eventually discharged to the Verde River at Childs, 4 miles above its confluence with Fossil Creek. A small reservoir (Stehr Lake) located between the power plants at Irving and Childs provides a 3-day supply of water to Childs when the diversion system in the upper watershed is shut down.

The power plants at Irving and Childs have a combined generating capacity of 4.2 megawatts and produce 37,000 mwhrs per year, enough power to sustain about 4,000 homes. The Irving plant, constructed in 1915, is located next to Fossil Creek and 4 miles downstream of the diversion dam. It produces one third of the power generated by the two plants. The Childs plant, constructed in 1908–1909, was one of the first hydroelectric power plants built in the West. It is located on the banks of the Verde River and generates the remaining two thirds of the power. Total generating capacity of these plants is less than 0.1 percent of the total power production capacity of APS. The Childs/Irving project was designated as a National Historic Mechanical Engineering Landmark in 1976 and was entered into the National Register of Historic Places in 1991.

Generation of the power produced by the Childs/Irving plants requires diversion of all of the natural discharge of Fossil Springs. Save for leakage of 0.2 cfs at the diversion dam and discharge of an additional 2 cfs at the Irving plant, the entire baseflow of the creek is contained within the water delivery system of the power project. Only flood flows and runoff in excess of the capacity of the diversion system (48 cfs) are allowed to flow down the natural channel.

Childs/Irving was issued a license for a period of 50 years on January 1, 1945 by the Federal Power Commission. In 1994, APS applied to the Federal Energy Regulatory Commission (FERC)

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for a new permit. Since 1990, the Forest Service, Arizona Game and Fish Department, and U.S. Fish and Wildlife Service have worked closely as an interagency team with APS to prepare the relicensing application. The team's sole function was to advise the applicant on affected resources. The application developed by APS does not consider alternatives for operation of the power plants and diversion works or for maintaining various flow scenarios in the creek. The Coconino and Tonto national forests are now conducting a joint environmental analysis with FERC that does consider alternative streamflow scenarios.

The entire relicensing process takes about 5 years and is triggered by the Federal Power Act that requires water power operators to be periodically reevaluated so that, if warranted, the operation may be either discontinued or modified, to reflect changing societal values, operational advancements, or other factors.

The Ecosystem

Vegetation on the upland watershed consists of ponderosa pine forest in the headwaters, followed by several plant communities including chaparral brushlands, pinyon–juniper woodlands and semi-desert grasslands as the creek descends. Elevations range from 7260 feet near the headwaters to 2550 feet at the Verde River confluence. Average annual precipitation is 18.1 inches at Childs and 19.9 inches at Irving.

The constant discharge of 43 cfs from Fossil Springs represents slightly more than 70 percent of the average annual water yield of the basin above the springs. A diverse aquatic and riparian community has developed downstream of the springs as a result of this reliable and sustained flow. The riparian community is diverse in both species composition and age structure. Woody riparian species include Arizona walnut, Arizona sycamore, velvet ash, Arizona alder, Fremont cottonwood, and willow species. The riparian community most closely resembles that of the Arizona sycamore–velvet ash type and appears to be typical of flood-dominated riparian systems in the southwestern United States. Seedlings are by far the most common age class and are concentrated in a narrow band along the fringe of the streams' free water surface. Older age classes are more prevalent on terraces further removed from the active channel of the stream. The proximity of seedlings to the stream results in high seedling mortality due to scouring from flood flows. Reduction in baseflow to 0.2 cfs below the diversion dam and to 2 cfs below the Irving plant restricts seedlings to a narrower band within the active channel than probably occurred historically when baseflow was 43 cfs.

The aquatic community consists of populations of predominantly native fish species. Above Irving Dam, roundtail chub, desert and Sonora suckers, and speckled dace are naturally present, and razorback suckers were stocked in 1988. Below Irving Dam, the first three species are still present, but in greatly reduced numbers due to reduction in flow. Longfin dace are also found in isolated reaches. Non-native species found below the diversion include green sunfish; smallmouth bass and yellow bullhead occur only in the lower 3 to 4 miles of the creek and probably originate from the Verde River. The razorback sucker is an endangered species, and the remaining native fish species have category 2 status (listing as threatened or endangered may be appropriate, but data substantiating the listing are not available). Razorback suckers are not native to Fossil Creek although they are native to Arizona. It is not known whether this population will become self-sustaining. The roundtail chub is the species of greatest concern in Fossil Creek because of its status under the Endangered Species Act and with both the Forest Service and Arizona Game and Fish Department. The Sonoran and desert suckers are also of concern because of declining populations throughout their range.

The large and sustained baseflow, the presence of a nearly intact native fish community, and a diverse riparian community all contribute to the remarkable resource values of Fossil Creek. But the factor that makes Fossil Creek unique is that the water discharged from the springs is laden with calcium carbonate that precipitates to form travertine deposits when exposed to atmospheric conditions. Travertine deposits are rare in Arizona and are considered to be outstanding natural resources where they occur (e.g. Havasu Creek, Tonto Natural Bridge, and the Little Colorado River).

The waters of Fossil Springs emerge from limestone, which provides a geologic environment conducive to the formation of travertine. Travertine is calcium carbonate deposited from solution in ground and surface waters. Chemically, travertine is identical to the mineral calcite. Water discharged from the springs has

high concentrations of calcium carbonate and dissolved carbon dioxide. As this water travels downstream and is exposed to the atmosphere, carbon dioxide gas is released. Release of carbon dioxide raises the pH of the water, causing the water to become supersaturated with calcium carbonate. When a critical level of supersaturation is reached, travertine precipitates from solution and deposits on the bed and banks of the channel. When the concentration of calcium carbonate falls below the critical level of supersaturation, formation of new travertine ceases.

Travertine deposition is influenced by external environmental factors as well as the chemical composition of the water. Most travertine deposition occurs at and below areas of turbulence, where the greatest amount of carbon dioxide is released. Algae also plays a major role in travertine deposition. Although not fully understood, two mechanisms are thought to be responsible. First, algae consumes carbon dioxide through photosynthesis, thereby acting as a type of outgassing mechanism. Secondly, algae filters suspended particles of calcite and provides a matrix for calcite nucleation and precipitation. Whatever the mechanism, the contribution of algae to travertine formation is significant.

In free-flowing streams, travertine precipitates on rocks, leaves, and other objects in the channel. Encrustation of these features by travertine, forming "fossils," accounts for the origin of the creek's name. Typically, dams are formed that can build up to several feet in height and create deep pools. A sequence of several pools often form in a stairstep pattern down the stream channel. It is not uncommon for travertine formations to accrete several inches per year in areas of stream turbulence.

Historic accounts predating the construction of Childs/Irving report large travertine structures in Fossil Creek. In 1891, Charles F. Lummis described waters "so impregnated with mineral that they are constantly building great round basins for themselves, and for a long distance flow down bowl after bowl." In 1904, F.W. Chamberlain, naturalist aboard the U.S.S. Albatross, reported dams "from several inches to a few feet in height, the highest is said to be 10 feet." He went on to describe the pools: "The largest pools seen were 50 to 60 yards long, 20 to 30 feet wide, and approximately 20 feet or more deep." According to Chamberlain's account, travertine formations were present along a 2-mile reach of the creek, beginning about a half mile below Fossil Springs. Today, relic travertine structures can be found at least 1 mile below Irving. Near Fossil Springs, massive travertine deposits 100 to 200 feet above the current stream channel are evidence that travertine must have formed in Fossil Creek for many thousands of years prior to construction of the power plants.

Management Issues

Both Forests view the relicensing and environmental analysis process as an opportunity to restore a unique ecosystem to Fossil Creek. From the Forest Service perspective, travertine and its importance to the ecosystem, particularly in terms of fish habitat and riparian resources, is the main issue driving the environmental analysis and derivation of alternatives.

Operation of the power plants reduces baseflow from 43 cfs to 0.2 cfs in the reach from the diversion dam to the Irving plant and from 43 to 2 cfs in the reach below Irving. Over the 85-year diversion period, travertine deposits that historically existed in the channel above the Irving plant have been eroded, primarily from flood flows. Formation of new travertine structures and growth of existing structures has ceased due to diversion of spring flows.

In the existing Fossil Creek system, travertine precipitates on the flume and tailraces of the hydroelectric facilities. However, even under the current operating system some formation of shallow travertine dams and structures is occurring below Irving where a minimum flow of about 2 cfs is maintained.

Construction and operation of the power plants changed the travertine balance of the system. Prior to construction of the power plant facilities, water flowed in the natural channel, and for about the first half mile downstream of the springs it had the appearance of a typical southwestern mountain stream with a moderate to steep gradient, confined channel and flood plain, and a gravel, cobble, boulder and bedrock streambed that generated significant turbulence and release of carbon dioxide. Beyond the first half mile below the springs, calcium carbonate reached supersaturation and travertine deposition occurred. Based on historic accounts, most if not all of the travertine was deposited in about a 2-mile reach below the diversion and above Irving. In the current system, water is diverted by the diversion dam just downstream of the springs into a relatively smooth-surfaced flume that generates little turbulence and provides lit-

tle opportunity for outgassing of carbon dioxide. Release of carbon dioxide does not begin in earnest under the current system until water is forcefully discharged from the tailrace at the Irving plant. As a result, most travertine deposition in the channel occurs below Irving and at rates much reduced from natural conditions. Significant deposition also occurs below the tailrace of the Childs power plant.

During the relicensing process the resource agencies and APS have been discussing the streamflow quantities necessary to maintain and enhance the natural resources associated with Fossil Creek. APS agreed that diversion of all flows from the creek to the power plants was not in the best interests of downstream resources. As a result, APS's application proposed to leave 10 cfs in the natural channel between the Irving Dam and Irving, and 5 cfs below Irving. Although this would be an improvement over existing conditions, Forest Service personnel did not believe that this flow regime would meet the objective of restoring this unique ecosystem. Accordingly, the Forest Service recommended to FERC that a range of alternative flow regimes, including full retirement of the project, be developed. Consequently Forest Service personnel are working with FERC to evaluate alternative flow regimes in the environmental analysis process. One flow regime being considered calls for restoring the entire natural flow of the creek between Irving Dam and Irving, the reach of Fossil Creek that historically had the greatest travertine deposition. This alternative would allow the travertine system to restore itself in the quickest manner possible, although it is not known how long this would take. Flows downstream of Irving would be increased to 5 cfs, an amount believed sufficient to allow the natural ecosystem to function, while still allowing power production to continue at Childs.

One of the major issues that developed during the environmental analysis process is whether travertine would reform in the stream channel under the reduced diversion alternatives. Discussions centered around the effects of watershed conditions on peak flows and the scouring effect of peak flows on travertine formations. One line of thought was that widespread overgrazing around the turn of the century so degraded watershed conditions that peak flows from storm events are much greater today than prior to the introduction of livestock. The increase in peak flows is thought capable of producing scouring floods that would destroy any potential new travertine formations.

It is well documented that large herds of cattle moved into southern and central Arizona in the late 1800s. These herds together with periods of drought resulted in the loss of much of the herbaceous and shrubby vegetation that for centuries provided protection for the soil. The net result was increased runoff and accelerated erosion. Eyewitness accounts of accelerated erosion during this period are well documented. One such account of Fossil Creek in 1904 (Chamberlain) states that "close pasturing" of ranges in the watershed probably resulted in the "unusual wash of mud that has made it more or less unfit for fish life." Since establishment of the Forests at the turn of the century, livestock numbers have been substantially reduced, and management practices improved. Soil conditions within the watershed have been mapped as a component of the Terrestrial Ecosystem Survey (TES) (Coconino National Forest 1994). The survey indicates that soil conditions are generally satisfactory, with current soil loss rates mostly within tolerable levels. Productivity of these soils is being maintained or improved due to acceptable vegetative ground cover that minimizes on-site soil loss.

Due to improvements in watershed conditions, the severity of runoff events is probably less today than at the turn of the century. Modeling of the Fossil Creek watershed, to evaluate peak flows under alternative ground cover conditions, suggests that peak flows under the most degraded watershed conditions were from 10 to 20 percent greater than under pre-livestock grazing conditions (Loomis 1994).

Even under natural conditions travertine systems are subject to regular cycles of deposition and erosion. During periods of heavy runoff, travertine formations can be destroyed by mechanical erosion due to abrasion by bedload materials and debris transported by flood flows. However, there is also evidence that travertine formations rebuild rapidly after such events. Havasu Creek below the rim of the Grand Canyon, approximately 150 miles north-northeast of Fossil Creek, is an example. It contains several miles of travertine-dominated dams and pools that are subject to periodic flooding. Recent severe flooding has impacted and removed sections of the travertine dams and deposits. However, the travertine quickly rebuilt and reestablished a system of dams and pools.

Historic observations of Fossil Creek also support the concept that travertine formations can withstand the increased severity of flood events attributable to degraded watershed conditions or will rebuild rapidly following such events. The presence of large travertine formations was reported by Lummis in 1891, and again by F.W. Chamberlain in 1904, long after heavy grazing had impacted the watershed but prior to diversions for Childs/Irving.

If we accept the idea that travertine formations will rebuild with maintenance of greater baseflows despite somewhat greater magnitude flood events, then based on the historical accounts of Lummis and Chamberlain, it is likely that with restoration of increased baseflows there would be a major change in the channel morphology of Fossil Creek above the Irving Plant. First, travertine appears to fill the interstitial spaces of the gravel and cobble deposits and acts as an adhesive to hold these deposits together. These "conglomerates" would have greater ballast than the individual particles and would be more resistant to movement during high flows. This stability is important for providing protection to looser deposits and rooting substrates. Secondly, travertine influences channel morphology through formation of the dams or bowls described in the Lummis and Chamberlain accounts. The presence of these dams results in a channel with a stepped profile, or as Lummis described it, "constantly building great round basins for themselves, and for a long distance flow down bowl after bowl." Formation of these structures would probably increase the stream's width and raise the water surface above the scoured bedrock channel that exists today. This in turn could increase bank moisture and possibly create a more lush riparian community than we see today. The steep sideslopes adjacent to the channel would probably prevent much increase in width of the riparian zone.

Travertine basins would also probably act as areas of deposition that would retain organic and inorganic materials in the system for a greater length of time for the biological community to utilize. Depositional sites are typically more productive than areas scoured to bedrock. In terms of fish habitat, formation of travertine dams in the low-flow channel would probably result in more habitat complexity than we see today. It is likely that a number of microhabitats would form in and around the dams, resulting in a range of water depths, velocities, and hiding cover. Today the habitat is in fairly large units of long boulder riffles and long bedrock-controlled pools with limited hiding cover and complexity. The complexity provided by travertine may also be important for providing areas of refuge during high flow events. Increased streamflow in other reaches of the creek would also benefit the predominantly native fishery simply by increasing the available habitat.

Travertine dams may also play an important role in flood plain function. By widening the channel and creating areas of flatter slope they help to dissipate flood flow energies. By creating depositional sites for riparian plants they create additional roughness elements that can also aid dissipation of flood energy. In today's system, many of the roughness elements resisting flood flows and holding the flood plain together are remnant travertine structures. Removal of the travertine formation process means that these remnant structures will slowly degrade over time through weathering and erosion. It is likely that the riparian community that has developed on flood plains supported by travertine structures will also degrade over time.

Riparian vegetation in stream reaches not affected by travertine deposition would also benefit from an increase in streamflows. Increased baseflow would move the area of seedling germination and growth farther from the low-flow channel than occurs under existing conditions. Exposure to scouring flood flows should be reduced and the rate of survival increased.

Ecosystem Management Considerations

The Forest Service mission is evolving under Chief Jack Ward Thomas to place greater emphasis on ecosystem management than has traditionally occurred within the agency. He believes the National Forest's traditional focus on commodity outputs should change "from sustained yields of products from ecosystems to the sustenance of ecosystems themselves" (Thomas 1994). He continues to support "the active use of ecosystems for the benefits they provide so long as that use does not unduly risk sustainability of those ecosystems." Application of the increased emphasis on ecosystem management to the Fossil Creek system would require restoration of increased baseflows to the natural channel. Restoration of this unique ecosystem would meet the intent of at least three

of the criteria listed in the Strategic Agenda in the Chief's Forest Service Agenda for the Future (Thomas 1994). These include:

1. Enhanced protection of ecosystems. This criterion emphasizes special care for ecosystems that are fragile or rare.
2. Restoration of deteriorated ecosystems. This criterion emphasizes repair or improvement of damaged ecosystems and management to improve the likelihood that diversity, long-term sustainability and future options are maintained.
3. Implementation of ecosystem management. This criterion emphasizes research and monitoring to help maintain the long-term health and productivity of ecosystems.

Throughout the relicensing and environmental assessment process (which is not yet complete), a major challenge has been working with other agencies and groups that have different missions, goals, and modes of operation. For example, the very rigid and strictly codified relicensing process of FERC limits adaptive management, and to a certain extent, interdisciplinary exchange. The process is focused on producing a license, and not necessarily a functioning ecosystem. On the other hand, the traditional multiple-use policy of the Forest Service encourages a "something for everyone" product, and again not necessarily a functioning ecosystem.

Agency cultures also played a large role in limiting discussions. For example the Forest Service has been primarily a rural-based, resource exploitation agency used to dealing with large-scale terrestrial resources in a pragmatic way, whereas FERC is primarily an urban-based, engineering-dominated agency used to processing (and approving) applications for hydropower development. APS clearly is in the business of producing and selling power, although their actions indicated a commitment to achieving a "greener" image. Other agencies, including the Arizona Game and Fish Department, U.S. Fish and Wildlife Service, Arizona Department of Environmental Quality, and APS's contractor also had different cultural and individual biases.

Dealing with these cultural biases and agency differences was always a challenge. Open recognition and discussion of differences was helpful, though not completely effective in breaking down barriers. Communicating a vision of a restored ecosystem was also a challenge. There was also substantial debate regarding the accuracy of technical information and the appropriateness of applying particular models to the ecosystems under study. Much of the modeling and analysis evaluated only the effects of changes in streamflow on the channel as it exists today and not as it could be in the future with the restoration of a travertine system.

Future analyses involving complex ecological relationships, interagency teams, or controversial issues could benefit from the experience gained during this process. To help visualize ecosystem potential, as opposed to current conditions, it is not only important to familiarize people with the area in question, but also with more pristine ecosystems having similar characteristics. Obtaining and communicating accurate technical information early in the process is critical. Once the initial information was gathered and analyses completed, these were accepted as "gospel" and were relied on throughout the assessment process despite criticism that revealed inadequacies and presentation of better information that became available later. It is also important to use models in the analysis that are appropriate for the area being studied, and to apply those models properly. Most importantly, it is vital to recognize that ecosystems and human societies are dynamic; a land use that was appropriate and needed at the turn of the century may be an anachronism in today's society.

Although the scientific and ecological relationships of the system are the focus of the current analysis, the final decision will be subjective and value based, with the human element playing a large role. The final outcome is still uncertain, but it is likely that base flows will be at least partially restored in Fossil Creek. Monitoring the changes in the ecosystem and the changes in human use of the watershed must be accomplished, and should be fascinating.

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An Ecosystem Management Strategy for The Sycamore Creek Watershed In South-Central Arizona

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The Sycamore Canyon watershed—a diverse, hilly ecosystem in the Pajarito and Atascosa mountains of south-central Arizona bordering Sonora, Mexico—is a perfect example of the USDA Forest Service theme, "Land of Many Uses." Unfortunately, many of these uses conflict and compete with each other for the same limited resources. Activities and designations on this 63 km^2 watershed include grazing of domestic animals, wildlife habitat, recreation, mining, roads, a research natural area, and a wilderness area. Alone and combined, some of these activities may cause soil erosion, channel silting, and the loss of habitat for rare riparian plants, amphibians, and aquatic species such as the Goodding's ash (*Fraxinus gooddingii*), the Tarahumara frog (*Rana tarahumara*), and the Sonora chub (*Gila ditaenia*). The Tarahumara frog has disappeared from the United States and the Sonora chub is state listed as endangered (Arizona Game and Fish Department 1988) and federally listed as threatened (U.S. Fish and Wildlife Service 1986).

The watershed has two distinct vegetation zones that separate near Hank and Yank Spring (Figure 1). The upper watershed ranges from 1,220 m to 1,667 m above sea level. The vegetation cover varies from oak woodland to oak grassland. Below the spring the watershed follows a narrow canyon almost to the Mexico border, where the elevation is 1,035 m. The vegetation on the lower watershed is mesquite grassland-Sonoran desert scrub ectone (Toolin et al. 1979). Sycamore Creek flows primarily in response to rainfall events, except in the lower watershed where it flows perennially in places to the Mexico border. There the canyon widens and the stream sinks into deep alluvial soils. The annual average precipitation in the area is 440 mm, recorded at the Nogales weather station. The annual mean temperature is about 24.7°C, ranging from 41°C in the summer to -13° C in the winter (Sellers et al. 1985).

The characteristics of the watershed that make it a focal point of human activity include its close proximity to the large metropolitan centers of Tucson and Nogales, a moderate climate, protected canyons with slick rock chutes and totem pole spires of rhyolite, and its intermittent water supply that supports a diversity of rare and sensitive riparian species unmatched within the Coronado National Forest, and perhaps throughout the southwestern United States. The flora of Sycamore Canyon includes 624 species of vascular plants, 20 species of lichens, and 40 species of mosses (Toolin et al. 1979). Today, at least 10 sensitive plants within the canyon are candidates for federal listing, including *Agave parviflora, Dalea tentaculodies*, and *Fraxinus gooddingii* (Warren 1993). Toolin et al. (1979) reported that Sycamore Canyon is the northern terminus for several Mexican plant species (e.g., *Henrya brevifolia, Acacia smallii*, and *Psoralea pentaphylla*) and at the same time is the southern limit in the range of some northern/montane species (e.g. *Amelanchier utahensis, Berberis wilcoxii*, and *Philadelphus microphyllus*). Bird watchers from across the country visit Sycamore Canyon to see five-striped sparrow (*Aimophila quinquestriata*), elegant trogon (*Trogon elegans*), and other rare species. The canyon also supports whiptail lizards (*Cnemidophorus burti* and *C. sonorae*), which are found only in southern Arizona (Peterson 1961, Stebbins 1987, Warren 1993). Sycamore and nearby Peñasco creeks are the only known waters with Sonora chub in the U.S. (Arizona Game and Fish Department 1988).

People are attracted to the watershed for its unique nature. Unfortunately, many of those that use the watershed add to its deterioration by hiking, driving, and camping outside designated areas. Other activities such as grazing and mining cause channel siltation and pollution of

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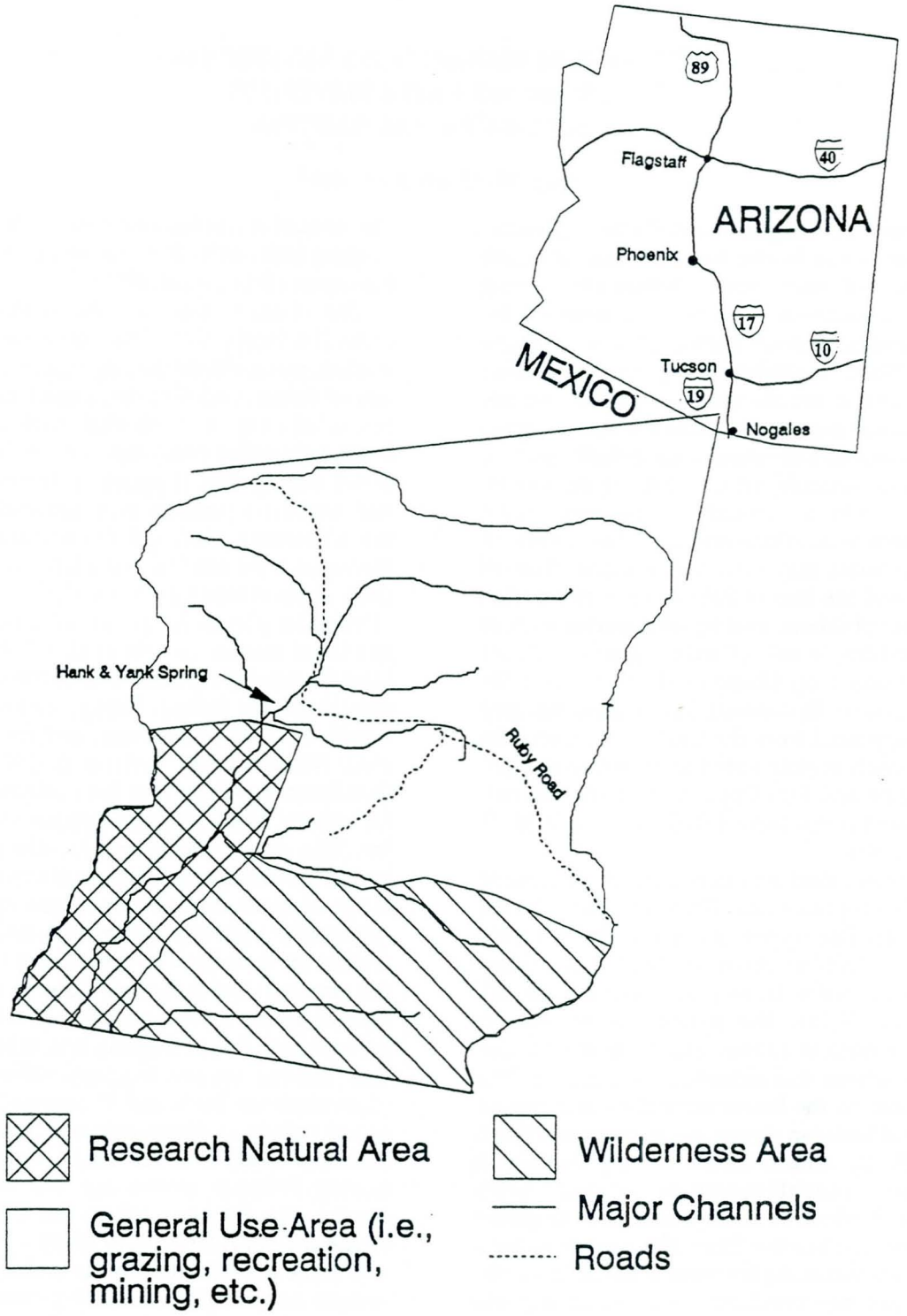

Figure 1. Location of Sycamore Canyon watershed.

aquatic habitats when practiced too close to channels and riparian areas. In view of the high potential for users to adversely affect this sensitive and unique ecosystem, the USDA Forest Service Nogales Ranger District needs to implement an ecosystem management strategy to protect the habitats of rare and sensitive plants, birds, and animals, and still permit ranchers, recreationalists, and others to enjoy the Sycamore Canyon watershed.

Background

In 1992 Dale Robertson, then chief of the USDA Forest Service, directed the agency to manage all forests and rangelands by "ecosystem management" (Gerlach and Bengston 1994). Under this policy, the Forest Service was to manage by a concept of "ecosystem sustainability" described as "the ability to sustain diversity, productivity, resilience to stress, health, renewability, and/or yields of desired values, resource uses, products, or services from an ecosystem while maintaining the integrity of the ecosystem over time" (Bormann et al. 1994).

There are many concepts, ideas, and definitions of ecosystem management (Jensen and Bourgeron 1994; Kaufmann et al. 1994; Salwasser 1994; Gerlach and Bengston 1994). On February 10, 1995, the chief of the USDA Forest Service, Jack Ward Thomas, informed the Forest Service community that he had appointed a policy team and charged them to "come up with definitive recommendations on how the federal government can take the lead in ecosystem management given its current resources" (Thomas 1995). In this paper, the following ecosystem definitions of Borman et al. (1994) are used:

Ecosystem management is a system of making, implementing, and evaluating decisions based on the ecosystems approach, which recognizes that ecosystems and society are always changing.

Ecosystem approach is a "system" in ecosystem that embodies three fundamental concepts: designating the physical boundary of the system and its parts, understanding the interactions of the parts as a functioning whole, and understanding the relation between the system and its context. Context means both the external factors that influence the system and also internal information that must be synthesized to be understood at the scale of the defined system.

Ecosystem sustainability is the degree of overlap between what people collectively want—reflecting social values and economic concerns—and what is ecologically possible in the long term (Figure 2). The overlap is dynamic because societal values and ecological capacity continually change. The desires of future generations can be protected by maintaining options for unexpected future ecosystem goods, services, and states.

Numerous ecosystem management efforts are in progress around the United States even as the USDA Forest Service team works to come up with definitive recommendations (Jensen and Bourgeron 1994; Richardson 1994; USDA Forest Service 1992). One example of a successful effort is on the Hoosier National Forest, where managers from the forest and researchers from the North Central Experiment Station developed a GIS-based harvest allocation model enabling managers to assess harvest strategies on forest structure and bird populations as far as 150 years into the future (USDA Forest Service 1995). Another example is on the Ouachita National Forest, where forest managers and researchers from the Southern and Southeast Experiment Stations are working to develop ecologically viable silvicultural techniques for regeneration of shortleaf pine/hardwood forests incorporating public involvement in the planning and decision-making process (USDA Forest Service 1995b).

Each ecosystem management effort differs in management objectives but the challenges that they must overcome are similar: (1) coordination across established administrative borders, institutional cultures, and other differences; (2) coordination across time horizons different from and longer than those of conventional decision making; (3) holistic coordination of solutions to reduce the likelihood that solutions to one problem will cause new problems; (4) decisions based on ambiguous and uncertain information; (5) public support and participation; (6) changes in rights and duties regarding natural resources; (7) fair distribution of costs and benefits of natural resource use; (8) sustainable development; (9) institutionalization of interdependence democratically; (10) conflict management; and (11) integration of human and biophysical factors (Gerlach and Bengston 1994).

Currently, Forest Service land managers develop and implement management practices

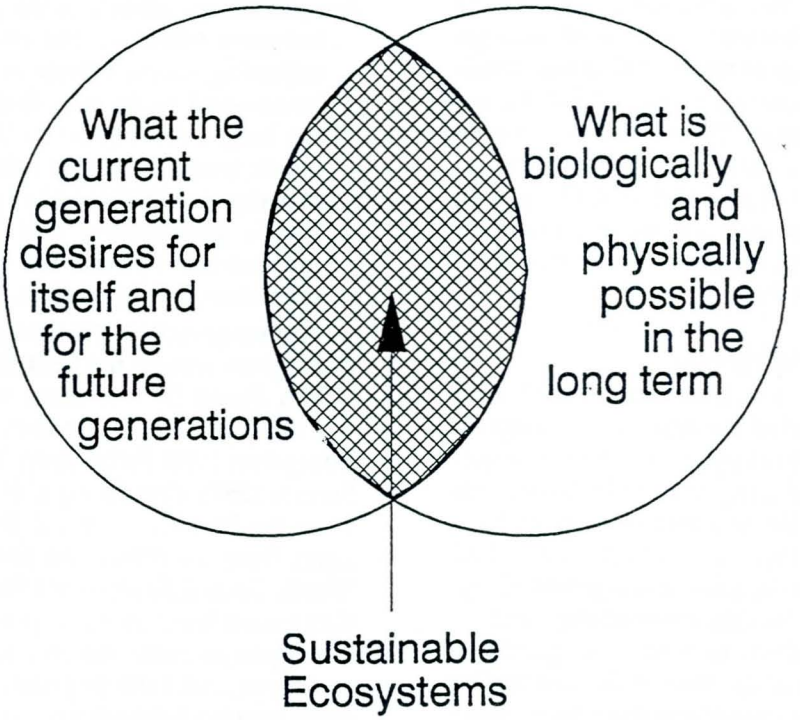

Figure 2. The elements of ecosystem sustainability (from Bormann et al. 1994).

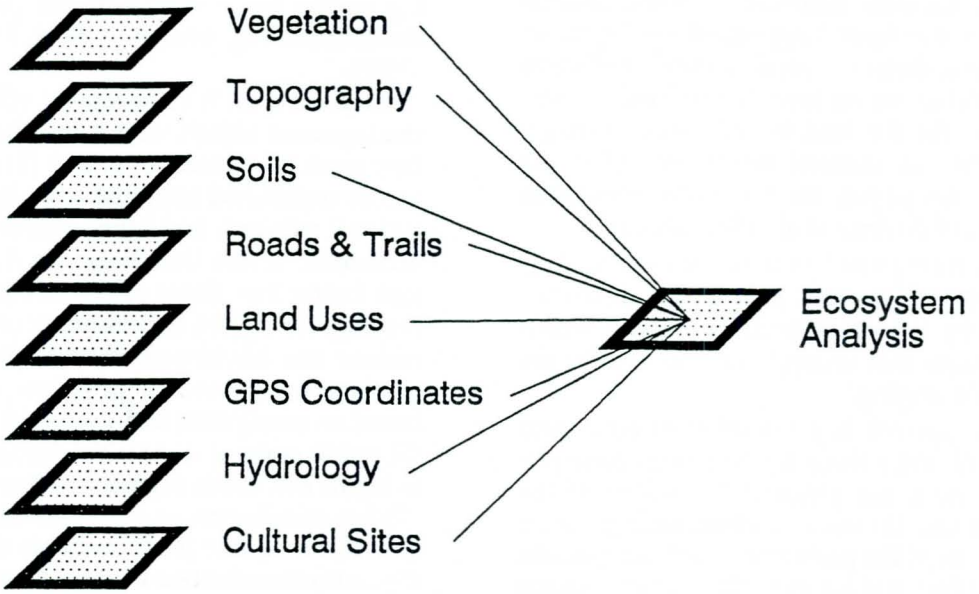

Figure 3. GIS elements of an ecosystem management analysis.

to provide sustainable opportunities including recreation, timber, research, and grazing for forest users while protecting the natural resources of watersheds. Practices and policies are often developed in conjunction with the United States Fish and Wildlife Service, the Bureau of Land Management, the Nature Conservancy, and other federal, state and private agencies, with some public participation and review.

Ecosystem management plan development, review, and updates include collection, analysis, and presentation of large data sets, reports, maps, slides, and videos. This process incorporates all the players of the traditional management plan development process, except now they will be called on to be more active participants. Depending on the project, this process can take weeks, months, or years to complete. Often times, duplication of previous data entry, reports, and manually produced work will take place. Finally, a thorough objective review of the finished product can be overwhelming, if not impossible, simply because of the quantities of materials produced.

Geographical information systems (GISs) can accelerate and clarify, through visualization, ecosystem management plan development, review, implementation, tracking, and updating. A GIS is a computer-based software program that allows aerial photos, topographic maps, text, satellite images, global positioning satellite (GPS) coordinates and other forms of data to be organized, analyzed, stored, and presented in graphical formats to help managers, decision makers, and other interested parties understand how ecosystem parameters and management options interact (McLean 1995; Naiman and Decamps 1991; USDA Forest Service 1994). Past, current, and projected data (soils, vegetation, topography, land uses, wildlife habitat, and climatic conditions) can be incorporated to create data layers that are used to make composite images and data summaries that display the interactions between ecosystem parameters (Figure 3). Image combinations are only limited by having the information needed to make informed decisions, cost, and the number of layers that can be interpreted and presented clearly together (McLean 1995).

Plan Development and Implementation

The Sycamore Canyon watershed offers an excellent opportunity for the Coronado National Forest to develop and implement an ecosystem management plan in an area where needed. Located 40 km west of Nogales and 93 km south Tucson with a regional population in excess of one million people, the area has the potential to be irreversibly degraded, leading to the loss of many rare and sensitive species, if an ecosystem management strategy is not implemented soon. Current land uses and designations are summarized in Figure 1.

The development and implementation of a GIS-based ecosystem management strategy for Sycamore Canyon watershed by the Nogales Ranger District (NRD) will require district personnel to pool their efforts and resources. Wildlife biologists, hydrologists, range conservationists, forest engineers, soil scientists, recreation specialists, and others will need to provide the data and information for and help create GIS data layers in their areas of expertise. The Forest resource staff has and/or collects much of the information required to develop the basic data layers of a GIS-based ecosystem management system (USDA Personal Communications 1994; Goldman 1993). These data and information include soils, vegetation, roads, infrastructure, land uses, topography, land ownership, cultural resources, and grazing allotment management plans. There have also been a number of studies and reports done on different aspects of the watershed vegetation (Toolin et al. 1979), aquatic habitats (Carpenter 1992, 1993), and geology (Riggs 1985) that can provide valuable information. Some of the information required to create data layers or completed layers may be available to the Forest from other agencies, such as the distribution of rare species from the Arizona Game and Fish Department, topography and geology from the U.S. Geological Survey, land ownership and resources information from the Arizona Land Department, and the distribution of riparian habitats from the Arizona Department of Water Resources. The public should also be considered in this process, so that their needs, desires, contributions, and influences on the Sycamore Creek ecosystem can be taken into account.

A GIS administrator (applications programmer and/or database specialist) should be designated to work with resource managers to ensure that data are collected, formatted, and stored on the computer system in appropriate formats for maximum use. In addition, the system administrator must work with the resource teams to understand the types of information (composite

maps, data summaries, etc.) needed to make their management decisions, and then to develop applications (macros) to query the GIS databases for this information. The resource management teams and the system administrator should collectively determine the look and feel of the GIS user-interface so that everyone is familiar and comfortable with the system. It should always be kept in mind that the GIS is a tool that is only as good as the data put into it and the macros created to analyze, combine, and extract information from the databases.

Composite images, data conversions and summaries, and other GIS outputs can help the Nogales Ranger District resource managers more effectively identify opportunities and locations of potential land use conflicts, soil erosion, non-point source pollution, recreation, habitat protection, and infrastructure. Depending on the data available, past, present, and future site comparisons and projections can be made, allowing resource managers to compare potential outcomes of their management recommendations in advance. An application that Forest personnel have already recognized as a time-saving use of GIS is locating roads, trails, recreation areas, range improvement sites, and OHV loading/off-loading sites (Goldman 1993). The system could be queried for potential sites based on user specified criteria (e.g., access, slopes, ownership, and proximity to rare and sensitive species and other selected resources). The outputs from the query could include maps showing potential locations, acreage summaries, current uses, and site conditions. Another application of the system is locating the sources of sediments that are silting in sections of the Sycamore Creek channel, near the Ruby Road crossing (Figure 1). In this case the output could be a composite map of the channel sections in question, soils, topography, and land uses. A general but effective use of the system would be easy storage, access, and updating of all forest resource information including mining claims, grazing allotments, archaeological sites, and other special use areas.

The hardware and software needed to develop and operate a GIS-based ecosystem management system will depend on the degree to which the forest elects to become involved. A basic system could be developed and implemented on a personal computer (PC) with a 486/66 or better computer chip, 8 megabytes of RAM (random access memory), and a 500-megabyte hard drive for general storage. Peripherals should include a color printer and a digitizer. Hardware purchases should be considered only after deciding upon the GIS software to be used for the project. There are numerous sources of hardware, software, and system development information (McLean 1995; USDA Forest Service 1994). The forest can also get recommendations from other national forests and government agencies with GIS technology in place.

Summary and Conclusions

The Sycamore Canyon watershed in south-central Arizona is a "Land of Many Uses" that conflict and compete for the same limited resources. Uses include grazing of domestic animals, wildlife habitat, recreation, mining, a research natural area, and a wilderness area. In addition, this area is home to numerous rare and sensitive riparian plants, amphibians, and aquatic species. The integration of these activities is degrading the ecosystem and causing the loss of habitat and species such as the Tarahumara frog which has disappeared from the U.S.

Ecosystem management and GIS can provide a method by which the resource managers of the NRD can manage this watershed with a holistic approach. Ecosystem management recognizes that all elements of an ecosystem are interconnected and should therefore be managed together, rather than concentrating on one theme such as aquatic habitat, grazing of domestic animals, or recreation. GIS is a computer software tool that permits image and information integration, management, analysis, and presentation through visualization. The combination of ecosystem management and GIS is very natural. Macros developed to query the GIS databases can facilitate resource managers' selection of program sites, tracking of land uses, and identification of problem sources, such as the origin of materials causing channel siltation in Sycamore Creek.

The metropolitan areas of Tucson and Nogales are rapidly expanding, and the Sycamore Canyon watershed is a focal point for recreationalists from these areas because of its close proximity and unique riparian plant, animal, and physical features. If the NRD is going to protect and preserve the habitat for rare and sensitive species of this ecosystem into the future, a GIS-based ecosystem management strategy should be developed and implemented now, before more species are lost.

Acknowledgments

We would like to thank Candis Allen, District Ranger; Tom Newman, Wildlife Biologist; and Barry Imler, Range Conservationist of the Nogales Ranger District, for their generous giving of time and information during our exploration of the Sycamore Canyon watershed.

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USING GIS AND REMOTE SENSING TECHNIQUES TO ESTIMATE LAND COVER CHANGES IN A DESERT WATERSHED

Diego Valdez-Zamudio[1]

Geographic information systems (GIS) and remote sensing techniques are powerful tools in the analysis of temporal changes in land cover or land use. Land cover refers to the type of feature present on the surface of the earth, including vegetation and nonvegetation features. Urban buildings, lakes, dense forests, and glacial ice are all examples of land cover types. Land use describes the human activity associated with a specific part of land and usually emphasizes the functional role of that land for economic activities (Lillesand 1987; Campbell 1987). Changes occurring in land cover and land use can generally be attributed to either natural or anthropogenic forces. Natural changes include both seasonal and annual variations in climatic conditions, and are often reflected by variations in natural land cover; fire can also induce natural changes. Changes resulting from anthropogenic forces are the result of human modification of the environment (Pilon et al. 1988). It is important to mention that the overall land use classes may change little over long periods of time, but the impact on wildlife populations and other components of affected ecosystems can be quite significant (Green 1994). GIS and remote sensing techniques can compare spatial information from two or more time intervals more readily than non-computer-based methods. Since the earliest Landsat imagery appeared in 1972, paired images from subsequent dates have been used to detect land cover and land use changes in the landscape (Iverson and Risser 1987).

Land use patterns in the study area, which is in the Sonoyta River watershed, are changing in both nature and intensity. Agricultural expansion has increased from 2,168 hectares in 1978 to nearly 10,000 hectares in 1987, and the population of Sonoyta has increased in the same period from less than 10,000 to more than 15,000 (Bennett and Kunzmann 1987). This growth suggests that the demand for urban and agricultural water will continue to expand. The exploitation of the underground water table in the Sonoyta River watershed and the expansion of agricultural activities in this region are presenting a serious threat to the wildlife and vegetation that are dependent on the soil and water resources of this watershed. Thus, these anthropogenic activities affect the biotic resources of different ecosystems included in and/or adjacent to important biological reserves such as EL Pinacate National Park in Mexico, Organ Pipe Cactus National Monument (ORPI), and the Cabeza Prieta National Wildlife Refuge in the U.S.

Brown et al. (1983) did an inventory of surface water resources at ORPI and found that gains in surface water sources have occurred due to the creation of many artificial water sources, including stock tanks, sewage disposal ponds, and wells with their attendant watering troughs. However, important losses of water sources have also occurred. Reviewing historical information, they found that in the early 1930s there were permanent *cienegas* (marshes) located along the Sonoyta River just south of the monument, which indicates that there was an important source of underground water in the watershed to support surface water bodies. Thus, they concluded that "the current pumping of ground water for agricultural purposes has further lowered the water table. As a consequence, the Sonoyta River today is a greatly depleted remnant of the surface water resource it once was. Historically, the flow from the warm springs at Quitovaquito had a surface connection with the Sonoyta River; this is no longer the case." Carruth (1994) reports a water level decline in the Quitovaquito Springs area caused by pumping near the Rio Sonoyta, Mexico.

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Therefore, considering that a better understanding of changes in historical land use should provide additional knowledge about conditions in the watershed, the principal objective of this study was to estimate the land cover and land use changes that occurred in an area located in the Sonoyta River watershed between 1972 and 1992 using remote sensing and GIS techniques.

Methods

The 8595 km^2 study area is located in the northwestern part of Sonora, Mexico and southwestern Arizona between north latitudes 31° 22' 55.3" and 32° 05' 49.5", and west longitudes 112° 09' 30.5" and 113° 18' 58.3" (Figure 1). Its climate is dry with a mean annual temperature of 21.4°C, maximum temperature of 45.1°C, and minimum temperature of -3.3°C. The mean annual precipitation is 209.7 mm (SARH 1994).

Detection of change in land use class areas was accomplished using Landsat MSS data processed with the remote sensing program ERDAS version 7.5, produced by Erdas, Inc., and the IDRISI GIS program version 4.1 developed by Clark University, Massachusetts. The scenes used in this study were acquired from the EROS Data Center of the U.S. Geological Survey in Sioux Falls, SD. The scenes are dated from September of 1972 and October 1992. The subsets defining the 1972 and 1992 study areas were extracted from the original scenes using the ERDAS program and the satellite imagery processing equipment available in the Remote Sensing Center, Office of Arid Lands Studies, University of Arizona.

After geometric and atmospheric correction, the 1972 and 1992 images were geographically registered. The identical geographic coordinates

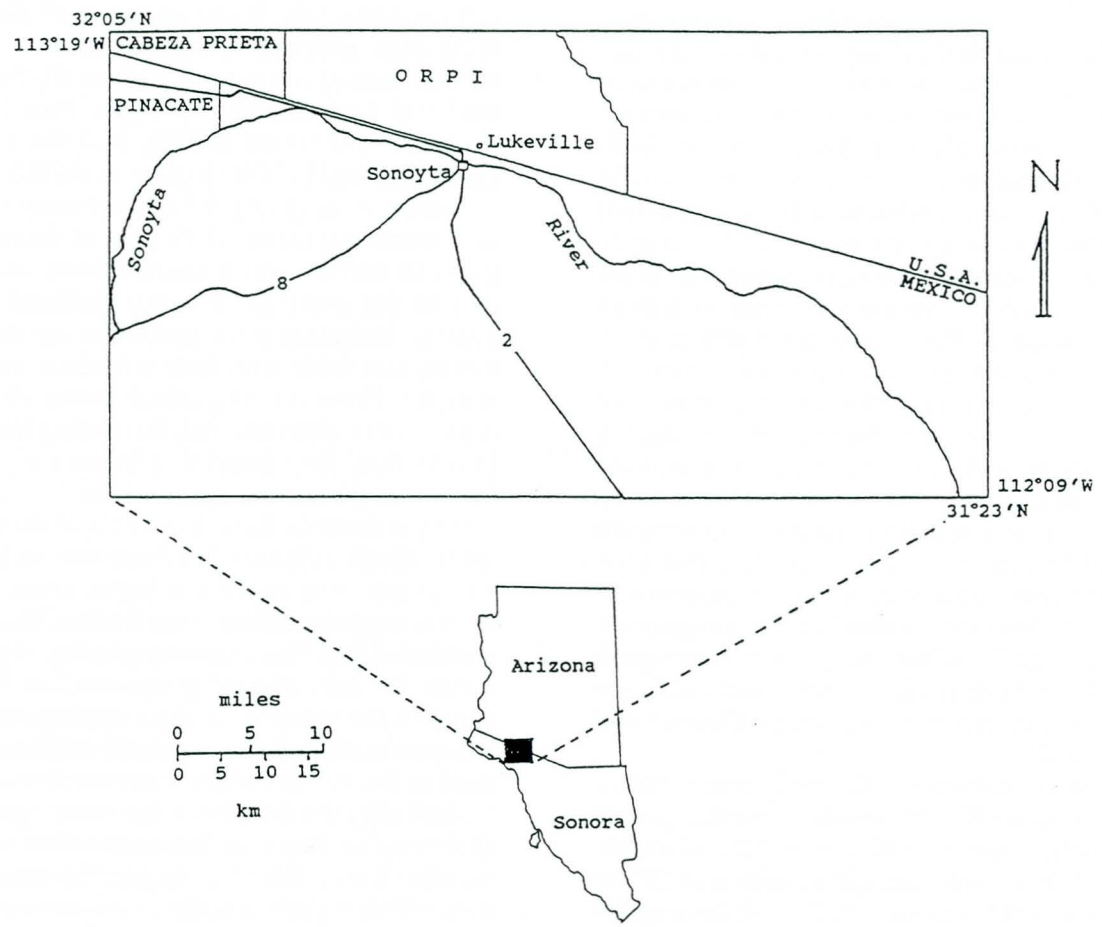

Figure 1. Extent of the study area.

of the two images allow overlay operations to compare geographic changes between the two dates of the images. With the intention of increasing contrast between different vegetation and other land cover classes on the 1972 and 1992 images, color composite images were created by combining bands 1 (0.5-0.6 um), 2 (0.6-0.7 um) and 4 (0.8-1.1 um) of the two images (Lillesand 1987).

To define changes between the two images, it was necessary to identify and classify the objects present on those images. By doing a preliminary unsupervised classification, six different land cover classes were adopted: croplands, bajadas with vegetation, plains with vegetation, riparian vegetation, hills, and bare soil. The choice of criteria used to distinguish these land cover classes was based principally on topography (slope), reflectance values, the association to other features on the image (like streams), and straight borders on irrigated fields. After two field trips to the site and revision of the ancillary data to obtain ground truth data, it was possible to identify training areas for a supervised classification.

The study used the classification method of seeding polygons because it involved small effort in the selection of the spectral values assigned to a land cover or land use class. For example, a polygon or border surrounding a known area of plains vegetation can be used to identify the variety of spectral values associated with this land cover class. These spectral values form the basis for selecting pixels with similar spectral values and assigning those pixels to the plains vegetation class. The training polygons should be as homogeneous as possible, containing only pixels that represent a particular land cover or land use class. The spectral signatures representing the desired classes were generated and the pixels of these signatures were sorted into classes using maximum likelihood classification decision rules (Erdas 1991). An error matrix of incorrect classification due to omission and commission was performed on the 1972 image. In accordance with Marsh et al. (1994), the error matrix analysis included errors of omission and commission, overall error measure, variance, and a Kappa index of agreement (KIA) (Eastman 1993).

Change analysis by image differencing (Jensen 1986; Campbell 1987; Singh 1989) was used to detect land cover and land use differences between 1972 and 1992. This method calculates the difference in reflectance values between two sets of Landsat data by subtracting the imagery of one date from that of another on a pixel by pixel basis. The subtraction process results in positive and negative values for pixels with reflectance value changes and zero values for pixels with no change. This procedure produces values for each band that are approximately Gaussian distributed, where pixels of no brightness value change are distributed around the mean and pixels of change are found in the tails of the distribution (Jensen 1986). In this study, change detection was performed by subtracting the 1972 classification image from the 1992 classification image.

Results and Discussion

Conspicuous differences in the extent of the land cover class areas can be seen between the two different years. Croplands appear as straight-bordered polygons. Bajadas with vegetation extend across slope gradients descending from hills to the river and other streams and to the valleys. Plains with vegetation are located in the valleys, between the bajadas and streams. The riparian vegetation class is typically found along the river and other streams located in the valleys, but it is also identified in the hill areas in the southern part of the image. Hills appear as an irregularly shaped, dispersed group of pixels that are defined by the pixels with the lowest reflectance values. The bare soil class is located mainly in the valleys and is defined by the pixels with the highest reflectance values.

For the years between 1972 and 1992, the classification method recorded 58.9 percent of the area as unaltered. Changes occurring in natural vegetation areas and not associated with obvious human activity affected 40.7 percent of the area, and less than 1.0 percent of the change was associated with human activities, primarily agriculture (Table 1). The positive change was in croplands, riparian vegetation and hills; but on the other hand, bajadas with vegetation, plains, and bare soil decreased in area (Figure 2). Among the six classes analyzed in this study, croplands was the least stable class: The classification process revealed that between 1972 and 1992 less than 4 percent remained unchanged. The most stable class was riparian vegetation.

Although the absolute percentage values for the fate of 1972 land cover and land use in 1992 were different in all cases, the relative ranking among the 1992 classes was identical for the fate

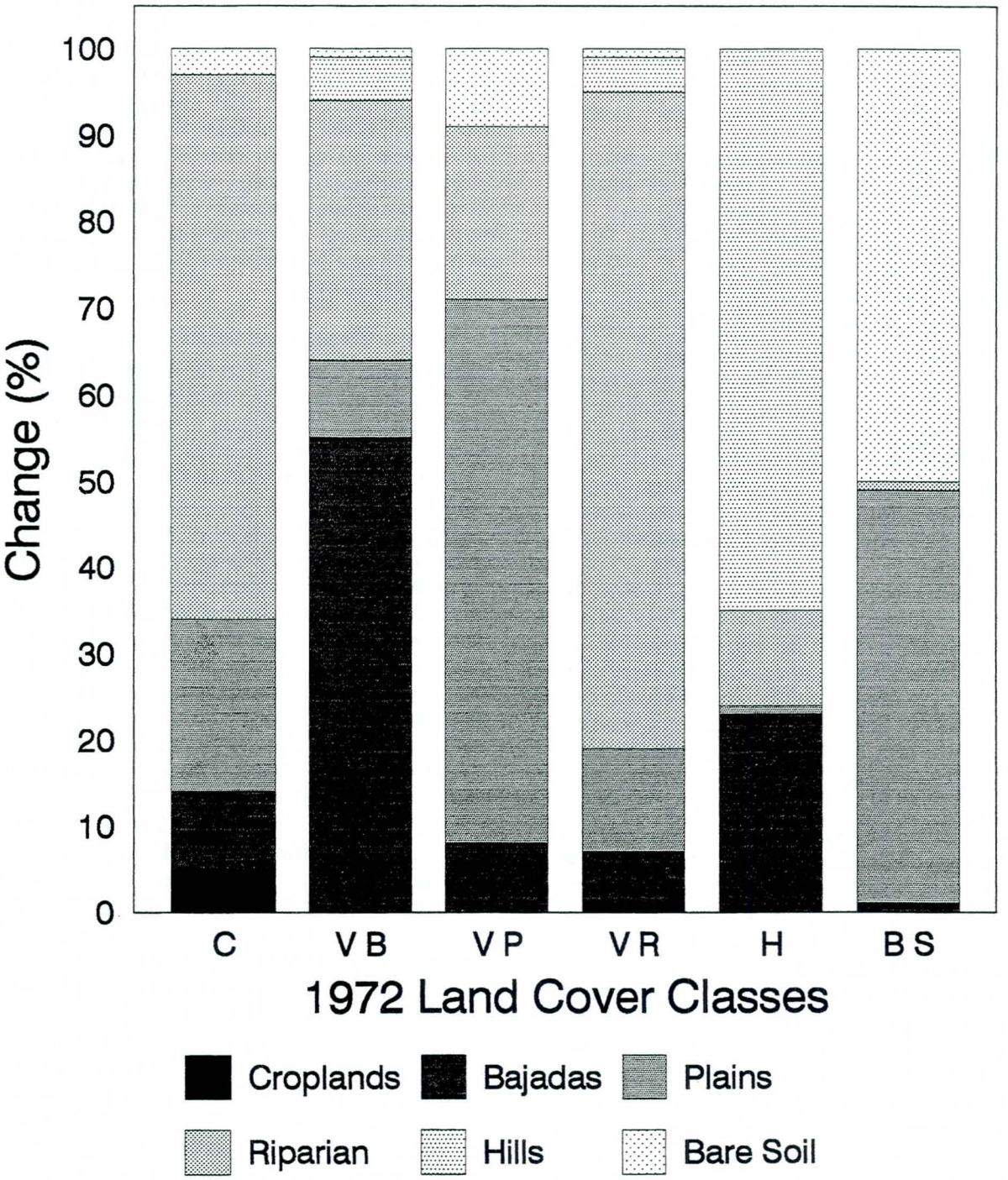

Figure 2. The fate of the 1972 land cover class in 1992 expressed as percent of 1972 total area after doing image classification by seeding polygons. C = croplands, VB = bajadas, VP = plains, VR = riparian, H = hills, BS = bare soil.

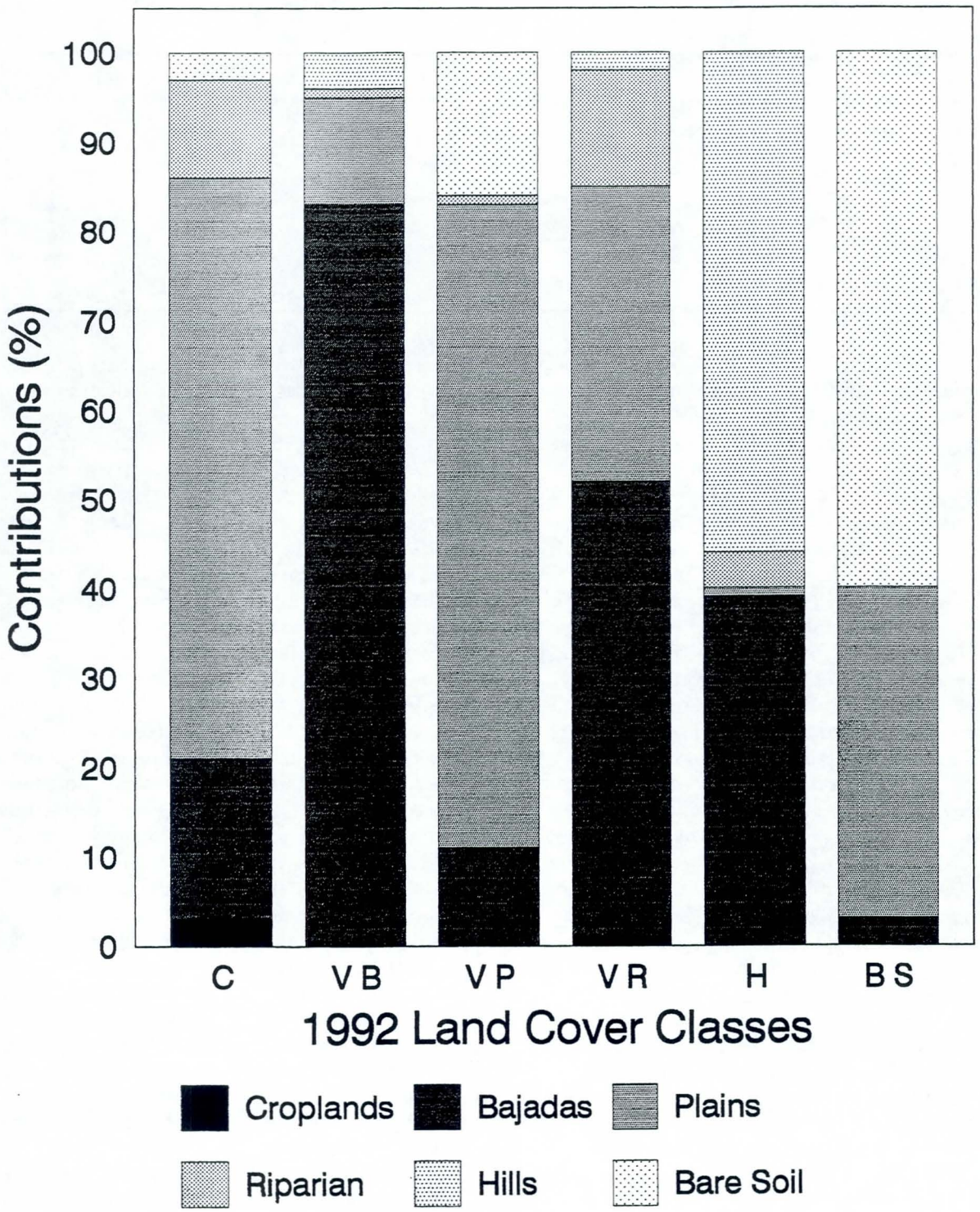

Figure 3. Source of the 1992 land cover classes expressed as percentage of 1992 total for each class using the seeding polygons classification method. C = croplands, VB = bajadas, VP = plains, VR = riparian, H = hills, BS = bare soil.

Table 1. Comparison of the source of 1992 land cover expressed as percent of 1992 total area for each class.

1972 Class	Croplands	Bajadas	Plains	Riparian	Hills	Bare Soil
Croplands	3.16	0.06	0.10	0.47	0.02	0.06
Bajadas	17.59	83.20	10.76	50.86	39.43	2.34
Plains	65.46	12.01	72.40	33.23	1.20	37.28
Riparian	10.80	0.97	1.44	13.29	3.54	0.27
Hills	0.13	3.45	0.10	1.86	55.78	0.06
Bare Soil	2.86	0.31	15.20	0.29	0.03	59.99
Total 1992	100	100	100	100	100	100

of croplands, bajadas, and hills. The classification method identified conversion to riparian vegetation as the most likely fate of 1972 croplands. The most likely fate of all other classes from 1972 to 1992 was to remain unchanged. Focusing on cropland fate, the classification method indicated that over 60 percent was converted to riparian vegetation and over 20 percent to plains vegetation.

The source of 1992 land cover and land use classes was less similar than the results for the fate of 1972 classes (Figure 3). The relative rankings among the 1972 classes as sources for 1992 classes were identical only for the hills class, and the greatest difference in the relative rankings was for croplands. Plains was the greatest source of 1992 croplands and bajadas was the greatest source of 1992 riparian vegetation. For all other classes, the greatest source of the 1992 extent was from the same class. Focusing on the source of 1992 croplands, plains contributed 65 percent and bajadas, the second greatest contributor, 17.5 percent.

Statistical analyses indicated no significant difference ($p > 0.05$) in annual precipitation and a significant difference ($p > 0.025$) for mean annual temperature between the periods before 1972 (1960-1971) and before 1992 (1980–1991).

The results of the image differencing process indicated that about 40 percent of the entire land cover in the study area changed between 1972 and 1992. Because desert vegetation is soon modified by climatic changes and considering that precipitation in the study area is the only factor of the environment that can cause a rapid change on vegetation (Cloudsley 1977), it is presumed that most of the changes were due to natural causes. Less than 1.0 percent of the area changed as a result of direct human activities such as agriculture. These anthropogenic changes occurred as a result of Mexican government policies to generate jobs and improve the regional economy of Sonoyta by financing the clearing of natural areas for agriculture and developing new wells to irrigate those new farmlands. Thus, the croplands changed from almost 300 hectares in 1972 to about 10,000 in 1992, and the number of wells increased from 4 in 1972 to 170 in 1992 (F. Mexía-Berúmen, personal communication; SARH 1989).

Cropland was the class with the highest rate of change, being principally reverted back into riparian areas by more than 60 percent, more than 20 percent into plains vegetation, and about 8 percent into bajadas. Croplands in 1972 were abandoned due to problems of salinity, erosion, and soil fungi diseases and the new farmlands are the result of conversion to replace abandoned farmlands and create additional farmlands (F. Mexía-Berúmen, personal communication). New croplands in 1992 were constituted principally by areas coming from riparian (11%), plains (65%), and bajadas (17%). This finding is probably evidence that former settlers in the study area used these particular land cover classes to develop their agricultural activities, and perhaps the same pattern of land selection is still being used by the new farmers. However, this does not mean that these classes can be totally converted into croplands due to some particular characteristics that make them unavailable for agricultural exploitation, including steepness, salinity, soil texture, and scarcity of water sources for crop irrigation. Although the anthropogenic changes obtained in this study do not represent a significant percent in terms of area, they are important in terms of environmental impact because they have generated a

considerable number of ecological problems including the depletion of the groundwater table; disappearance of former perennial water bodies and streams that were sources of water and habitat for wildlife and plant species; reduction on the wildlife population species in the area, modification and suppression of natural habitats and ecosystems, ecosystem pollution by extensive and intensive use of agrochemicals, and weed dispersion and introduction into natural adjacent ecosystems.

There are several techniques to classify satellite imagery, all of them differing in processing time, algorithms used, and accuracy yielded. In this study, image classification was performed using the seeding polygons classification method. This widely used classification method proved to be uncomplicated, was easy to understand and manipulate, and can yield good results with an acceptable level of accuracy in a relatively short period of processing time, which is a good characteristic desired in every classification method. The principal disadvantage of this method is that when polygons are not homogeneously drawn (when enclosing pixels for a given class), the number of errors of omission and commission will increase and the level of accuracy of that classification will be too small to be considered acceptable for satisfactory results. When this occurs, new operations have to be performed until good results are obtained. Thus, this method depends principally on the user's ability and eyesight to distinguish between pixels with very similar reflectance values. The seeding polygons classification method yielded an overall accuracy of 81 percent in both 1972 and 1992 images.

Accuracy tends to degenerate when grid-cell resolution approaches the image pixel dimension (Owe 1984). In this study, pixel size (57m x 79m) was big enough to include several dissimilar classes, each with its own spectral properties. A pixel's spectral intensities could indicate one of the contained classes or an average of all, resulting in an unrelated surface. Most of the more common errors are related to these spectral anomalies. This result could explain the fact that large areas in the hills appear with the same reflectance values as riparian vegetation zones along the Sonoyta River. Thus, under more homogeneous land use conditions, more regional applications, or using higher resolution satellite imagery, the seeding polygons classification method could yield more accurate results.

In comparison to other life zones, the desert displays changes in land cover that are not consistent and may only manifest recent variations in rainfall, and that may not be significant for a longer period of time (Hastings 1980). In environments like the study area, climatic variations from year to year are substantial, and as a consequence, a natural change can be evident, as occurred in these two dates studied.

There was a consistent difference between precipitation means (19.2), but this difference was small considering the variability of the measurements (STDEV1 = 76.6, STDEV2 = 99.2). The computed t-value (-0.53) has a p-value of 0.60, indicating very little evidence of a difference in precipitation between 1972 and 1992. However, extreme changes in precipitation pattern in some years (i.e. 31.9 mm in 1968 and 487.0 mm in 1983) could have affected the vegetation land cover classes, resulting in spectral differences between 1972 and 1992 images that could be interpreted as a natural change. For example, the wet condition in 1983 may have increased vegetation cover, resulting in less bare soil in 1992 than 1972.

Short time periods (i.e. each 5 years) are recommended for change detection in order to create simulation models of the change processes, and therefore have a better understanding of how different factors are interacting to cause class modification and predict possible changes in the future.

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A Stochastic Model for Simulating Daily Temperature and Humidity on a Ponderosa Pine Type Watershed

David E. Rupp and Aregai Tecle[1]

A stochastic model is constructed to simulate the daily maximum and minimum temperatures and the daily vapor pressure deficits throughout the cold season for a site in the ponderosa pine forest of the Beaver Creek watershed of north-central Arizona, where the cold season is defined as the period beginning on October 1 and ending on April 30. The synthetic temperature and vapor pressure data are to serve as inputs to a cold-season water yield model designed for use in upland watersheds.

In this study, the daily temperature is described as a function of the day of the season and a randomly generated variable. The temperature is also dependent on whether or not precipitation occurs. In turn, the vapor pressure deficit is described as a function of temperature that differs depending on whether the day is wet or dry. The occurrence of precipitation is described as a stochastic process in Rupp (1995). A detailed survey of stochastic daily weather simulation modeling is presented in Richardson (1981). Other more recent models includes those of Desanker and Reed (1991) and Racsko et al. (1991).

Though the simulation of temperature and/or humidity has not been done for the area under study before, temperature and relative humidity patterns in the ponderosa pine forests of Arizona have been analyzed by Beschta (1976) and Campbell and Ryan (1982). According to these studies, the mean monthly cold-season temperatures range from a low of approximately -2.5°C in January to 7.2°C in October, while diurnal temperature fluctuations average 17°C. Similarly, the diurnal relative humidity varies significantly during the cold season, with an average of 75 percent at 5 a.m. and an average of 45 percent at 5 p.m. Beschta (1976).

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Methods

The data used in this analysis were collected continuously during the years of 1962 through 1982 at a site in the ponderosa pine type forest of north-central Arizona. The collection site is located in the Coconino National Forest near the outlets of the Bar-M and Woods Canyon watersheds at an elevation of 1,945 m, a latitude of 34° 52' 04", and a longitude of 111° 36' 47". A hygrothermograph was used to record the air temperature and relative humidity. This study uses the collected data to examine the daily maximum and minimum temperatures and vapor pressures at the study site during the cold season.

Analysis of Temperature

From the data sets of daily maximum and minimum temperatures a set of diurnal temperature variation is created. The sets of daily maximum temperature and diurnal temperature variation are then divided into those days in which precipitation occurred and those days in which it did not. Different functions are fit to the wet and dry day data to describe the change in the mean daily maximum temperature and the change in the diurnal temperature variation throughout the cold season. An error term is used to describe the variations of the daily maximum temperature and diurnal temperature from their respective means.

During the simulation process, the daily maximum temperature and the diurnal temperature variation are generated independently. The daily minimum temperature is then calculated by subtracting the diurnal temperature variation from the daily maximum temperature.

One method of modeling the random nature of a variable that exhibits persistence from one day to the next uses a lag-one Markov process (Fiering and Jackson 1971). With temperature as

the random variable, the lag-one, multi-periodic Markov equation takes the general form

$$T_{i,j}=T_{ave,j}+r_j\frac{S_j}{S_{j-1}}(T_{i,j-1}-T_{ave,j-1})+t_{i,j}S_j(1-r_j^2)^{1/2} \quad (1)$$

where $T_{i,j}$ is the temperature generated on day j in season i, $T_{i,j-1}$ is the previous day's temperature in season i, $T_{ave,j}$ is the mean temperature for day j, $T_{ave,j-1}$ is the mean temperature for day j-1, r_j is the lag-one autocorrelation coefficient between the temperatures of days j and j-1, $t_{i,j}$ is a normally distributed random variable with zero mean and unit variance for day j in season i, and s_j and s_{j-1} are the standard deviations of the temperatures for days j and j-1, respectively (Fiering and Jackson 1971).

For this study, two separate forms of equation (1) are used: one for days in which precipitation is absent and the other one for those in which it is present. Dropping the subscript i from equation (1) because the simulation is done for one season—the cold season—only, and assuming that the autocorrelation coefficients and standard deviations for daily temperature are constant throughout the season, results in equation (1) becoming

$$T_j = T_{ave,j} + r(T_{j-1}T_{ave,j-1}) + t_jS(1-r^2)^{1/2}. \quad (2)$$

Two sets of autocorrelation coefficients and standard deviations are determined separately from the dry and wet day data sets.

The mean dry and wet daily maximum temperatures are varied with time using functions that are fit to the data. Specifically, parabolic functions used to describe the change in daily maximum temperature with time for dry days and wet days are, respectively,

$$T_{dry} = b_0 + b_1x + b_2x^2 \quad (3)$$

and

$$T_{wet} = b_3 + b_4x + b_5x^2 \quad (4)$$

where T_{dry} and T_{wet} are the average maximum daily temperatures for dry and wet days, respectively, x is the time in days since the beginning of the season, and b_0 through b_5 are regression coefficients. A least-squares regression procedure is used to find the best fit to the data. The standard deviations for the dry and wet temperatures used are the root mean square errors of the dry and wet day regression equations, respectively. T_{dry} and T_{wet} are substituted for T_{ave} in equation (2) depending upon the state of precipitation on a given day.

When a wet day follows a dry day, or a dry day follows a wet day, a special case of equation (2) for r_j equals zero is used. In other words, the model assumes no persistence for these two cases. This is analogous to saying that the arrival or departure of a storm system interrupts the prevailing temperature pattern completely, and the new day's temperature will be unaffected by the previous days temperature and will be solely a function of the mean and the probability distribution for the new day's temperature. When the autocorrelation is set to zero, equation (2) reduces to

$$T_j = T_{ave} + t_jS. \quad (5)$$

Finally, to simulate the minimum daily temperature, the diurnal variation is generated in the same manner as that for maximum temperature. The first step is to describe any trends in diurnal variation with time throughout the cold season. As with maximum temperature, parabolic functions are used to model the change in diurnal variation with time for both days of no precipitation as well as days with precipitation. The functions for dry days and wet days are, respectively,

$$DV_{dry} = b_6 + b_7x + b_8x^2 \quad (6)$$

and

$$DV_{wet} = b_9 + b_{10}x + b_{11}x^2 \quad (7)$$

where DV_{dry} and DV_{wet} are the diurnal variations for dry and wet days, x is the time in days, and b_6 through b_{11} are regression coefficients.

Simulation of the diurnal variation uses the same procedure as that of maximum temperature. The lag-one Markov equation for diurnal temperature variation is expressed as

$$DV_j = DV_{ave,j} + r_{dv}(DV_{j-1} - DV_{ave,j-1}) + ts_{dv}(1-r_{dv}^2)^{1/2} \quad (8)$$

where the diurnal variation (DV) has simply replaced the maximum temperature (T) in equation (2) and the subscript dv indicates that the autocorrelation coefficient and standard deviation are calculated from the diurnal variation data. As is done with temperature, two sets of autocorrelation coefficients and standard deviations are determined, one for the dry day data and one for the wet day data.

As with the simulation of maximum temperature, when a dry day follows a wet day, or a wet day follows a dry day, the diurnal variation is simulated by the special case of equation (8)

for r equal to zero. The equation for a dry day following a wet day or a wet day following a dry day is

$$DV_j = DV_{ave} + t_j S_{dv} \quad (9)$$

where the diurnal variation (DV) has simply replaced the maximum temperature (T) in equation (5). Calculation of the daily minimum temperature is made by subtracting the simulated diurnal variation, DV_j, from the simulated daily maximum temperature, T_j.

Analysis of Humidity

The relationship between the vapor pressure deficit and the air temperature at the study site during days in which precipitation occurs and during days in which it does not is examined. Vapor pressure deficit is chosen as the measure of humidity to be analyzed because it is a critical factor in estimating evapotranspiration. Values of vapor pressure deficit are estimated from the measured daily maximum temperature and daily minimum relative humidity.

The vapor pressure deficit, Δe, is the difference between the vapor pressure that the air would have if saturated, e_s, and the actual prevailing vapor pressure, e. The saturation vapor pressure can be estimated by

$$e_s = 6.11 \exp[17.3 T_a / (T_a + 237.2)] \quad (10)$$

where e_s is saturation vapor pressure (mb), and T_a is air temperature (°C) (Dingman 1994). The actual vapor pressure is determined from the saturation vapor pressure and the relative humidity using

$$e = e_s RH / 100 \quad (11)$$

where RH is the relative humidity.

The vapor pressure deficit, Δe, is calculated from the air temperature and constants empirically derived through a linear regression procedure for the study area. The equation developed for estimating the vapor pressure deficit given the air temperature is

$$\Delta e = a_1 \exp[17.3 T_a / (T_a + 237.3)] + b_i \quad (12)$$

where a_i and b_i are constants derived by regressing vapor pressure deficit to air temperature. The subscript i refers to two sets of a and b that are derived: one set for days in which no precipitation occurs and the other for days in which there is precipitation.

Results

Analysis of Temperature

Cold-season daily maximum temperatures for both the dry and wet data set are described using parabolic functions. Temperature is regressed to day of year to determine the equation that best fits the data. In constructing the regression equations, October 1 was set as day 1 and April 30 as day 212, the end of the cold season.

The regression equation for the dry daily maximum temperature with respect to the time of the cold season is

$$T_{dry} = 23.66 - 0.2668(DAY) + 0.001175(DAY)^2 \quad (13)$$

where T_{dry} (°C) is the estimated mean daily maximum temperature in the absence of rain and DAY is the number of days since the beginning of the cold season. The root mean square error (RMSE) of the regression is 4.74157, the value for R^2 is 0.4306, and the regression coefficients are all highly significant with p-values less than 0.0001. Similarly, the regression equation for the wet daily maximum temperature in terms of the days since the beginning of the cold season is

$$T_{wet} = 15.31 - 0.2164(DAY) + 0.000922(DAY)^2 \quad (14)$$

where T_{wet} (°C) is the estimated mean daily maximum temperature in the presence of rain. The RMSE for the regression is 4.3838, while the value of R^2 is 0.3104 and, as with dry maximum temperature, all regression coefficients are significant at the 0.0001 level.

The low R^2 values for both regression equations reflect the high degree of variability among daily maximum temperatures. Therefore, to better view the cold-season temperature trend for dry and wet days, the average dry and wet daily maximum temperatures are calculated for the 20 years of record and plotted in Figure 1. Superimposed on Figure 1 are plots for the maximum daily temperatures for dry and wet days using equations (13) and (14), respectively.

The same procedure described above is repeated for the diurnal temperature variation, or difference between the daily maximum and minimum temperatures. Again, parabolic functions are fit to the dry diurnal variation and the

wet diurnal variation. For the case of dry days, the best-fitting equation is

$$DV_{dry} = 22.33 - 0.06034(DAY) + 0.000273(DAY)^2 \quad (15)$$

where DV_{dry} (°C) is the estimated mean diurnal variation in temperature during days when precipitation is absent. The regression has an RMSE of 5.3475 and an R^2 value of 0.029. Even though the model explains little of the variability, the regression is significant at the 0.0001 level.

For the wet day data, the regression equation for the diurnal temperature variation with respect to days since the beginning of the cold season is

$$DV_{wet} = 11.30 - 0.050267(DAY) - 0.000244(DAY)^2 \quad (16)$$

where DV_{wet} (°C) is the estimated mean diurnal temperature variation during days when precipitation occurs. The RMSE is 5.6671. The value of R^2 is 0.0166, and though it is even lower than that for the dry day case, the regression equation is significant at the p-value of 0.0011. Similar to the maximum temperature data, the low R^2 indicates a high amount of variation. Figure 2 shows plots of the daily averages of dry and wet diurnal temperature variations for the cold season. Superimposed on them are plots of diurnal variation values generated using equations (15) and (16), respectively.

Simulation of the daily maximum temperatures for consecutive dry days and consecutive wet days using the Markov process in equation (2) requires determining the lag-one autocorrelation coefficients for each day type. The autocorrelation coefficients for both wet and dry days are found to be high: 0.890051 and 0.839985, respectively. This indicates that the daily maximum temperatures do not vary greatly from one day to the next.

Similarly, simulation of the daily minimum temperature using the Markov equation (8) requires the use of the lag-one autocorrelation coefficients for diurnal temperature variation for both consecutive dry days and consecutive wet days. The autocorrelation coefficient for consecutive dry days equals 0.507597, while for consecutive wet days it is equal to 0.279799.

To visually examine the performance of the temperature model, it is used to simulate one cold season of temperatures and the results are plotted and compared to one cold season of measured daily temperature data. Though the measured and simulated data cannot be compared on a one-to-one basis, general overall trends can be examined. Figure 3 shows the measured daily maximum and minimum temperatures for the 1962–1963 cold season, while the simulated cold season data are illustrated in Figure 4. As with the measured data, the simulated results show a general decrease in temperature as the cold season progresses, followed by a warming as the season concludes. In addition, the variations in the simulated daily maximum temperatures are comparable to the measured values. However, there appears to be more variation among the simulated daily minimums than among the measured minimums. This may be a result of simulating the minimum temperature indirectly via simulation of the diurnal temperature variation.

Analysis of Humidity

The vapor pressure deficit, Δe, is derived as a function of temperature for the study area. The equation developed for estimating the vapor pressure deficit during days in which no precipitation occurs is

$$\Delta e = 5.29\exp[17.3T_a / (T_a + 237.3)] - 1.69 \quad (17)$$

where Δe is the vapor pressure deficit and T_a is the air temperature. The value of R^2 for the regression is 0.934 and the RMSE is 1.512. On days in which there is precipitation, the equation for estimating the vapor pressure deficit is

$$\Delta e = 4.70\exp[17.3T_a / (T_a + 237.3)] - 2.90. \quad (18)$$

The regression equation has an R^2 value of 0.737 and an RMSE of 1.888. Figures 5 and 6 show scatter plots of vapor pressure deficit verses temperature for days of no precipitation and days of precipitation, respectively. Superimposed on the figures are plots of regression equations (17) and (18).

Summary and Conclusions

An analysis of daily temperature versus days since the beginning of the cold season reveals that cold-season temperatures can be described using parabolic functions fit to the data. However, the analysis reveals the existence of too much variability in the daily changes in temperature to describe them sufficiently using a single function. Therefore, the variability of daily temperature is modeled in addition to the daily trend. The variation in temperature from the parabolic regression curve is simulated using a

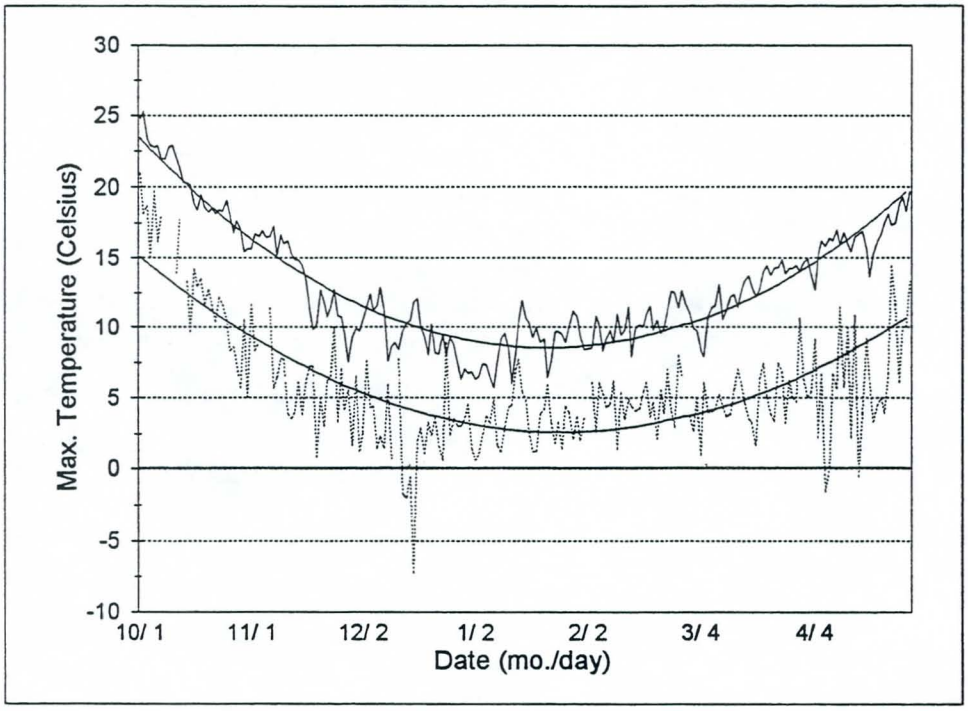

Figure 1. Average daily maximum temperature for dry (solid) and wet (dashed) days with parabolic functions.

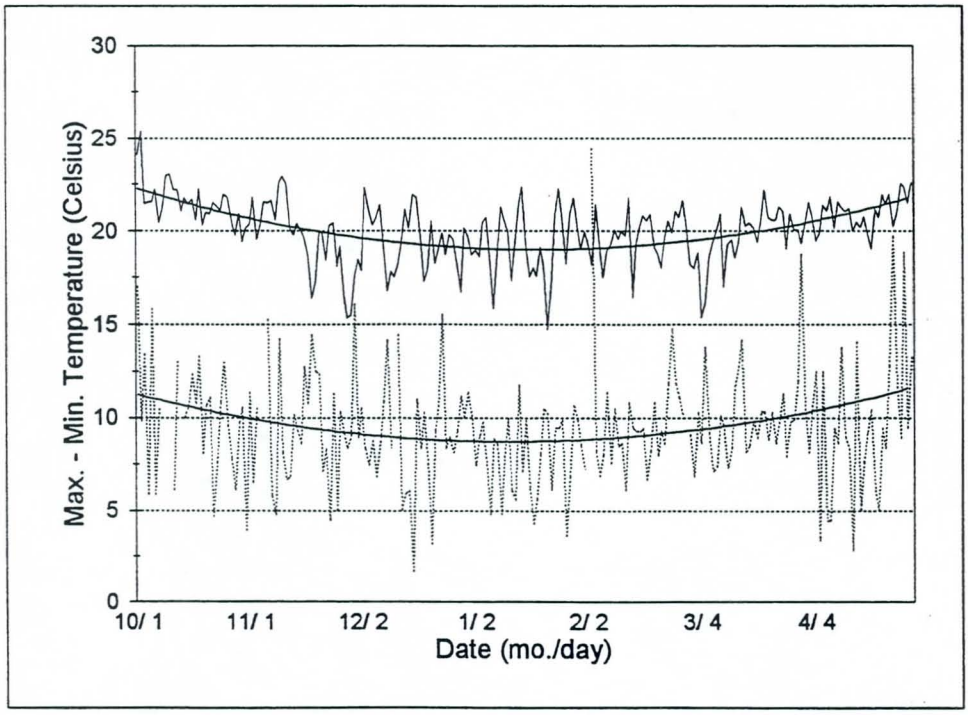

Figure 2. Average diurnal variation for dry (solid) and wet (dashed) days with parabolic functions.

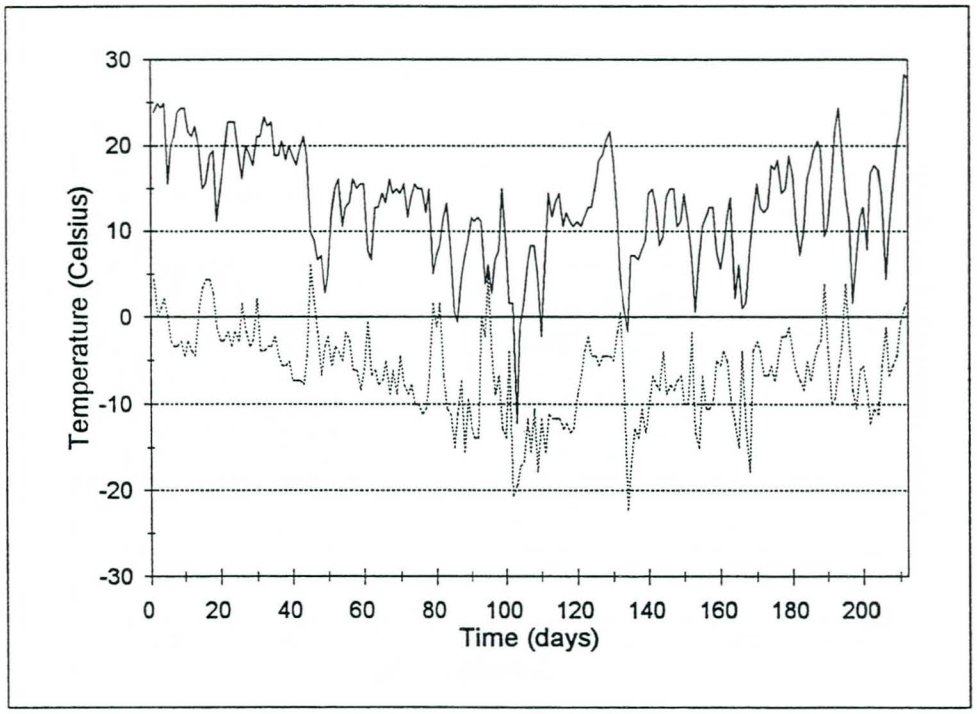

Figure 3. Measured daily maximum (solid) and minimum (dashed) temperatures for representative cold season.

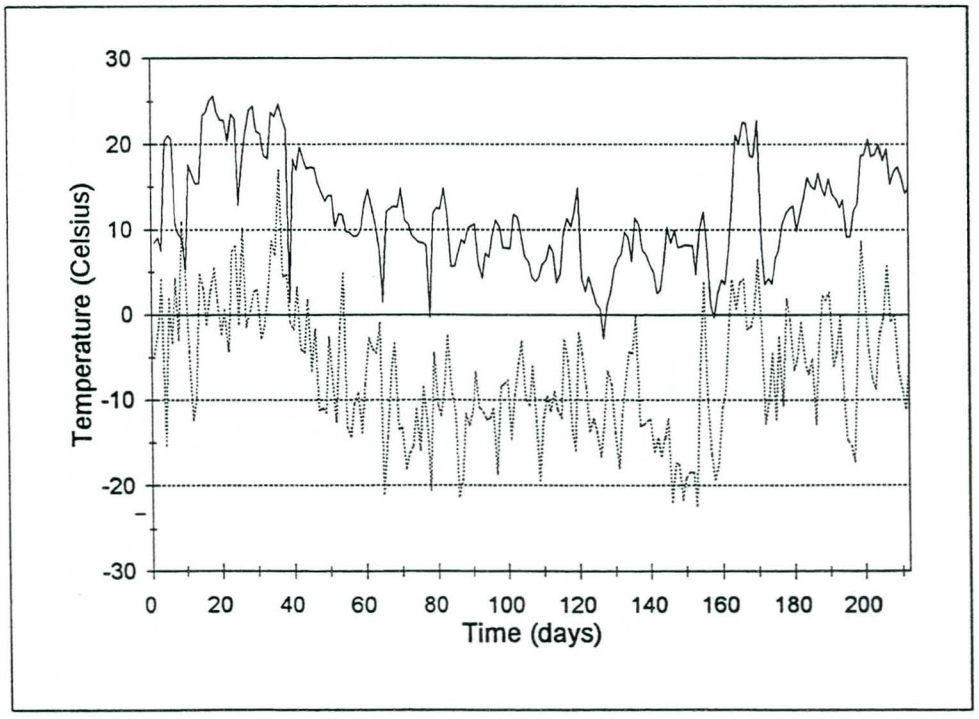

Figure 4. Simulated daily maximum (solid) and minimum (dashed) temperatures for one cold season.

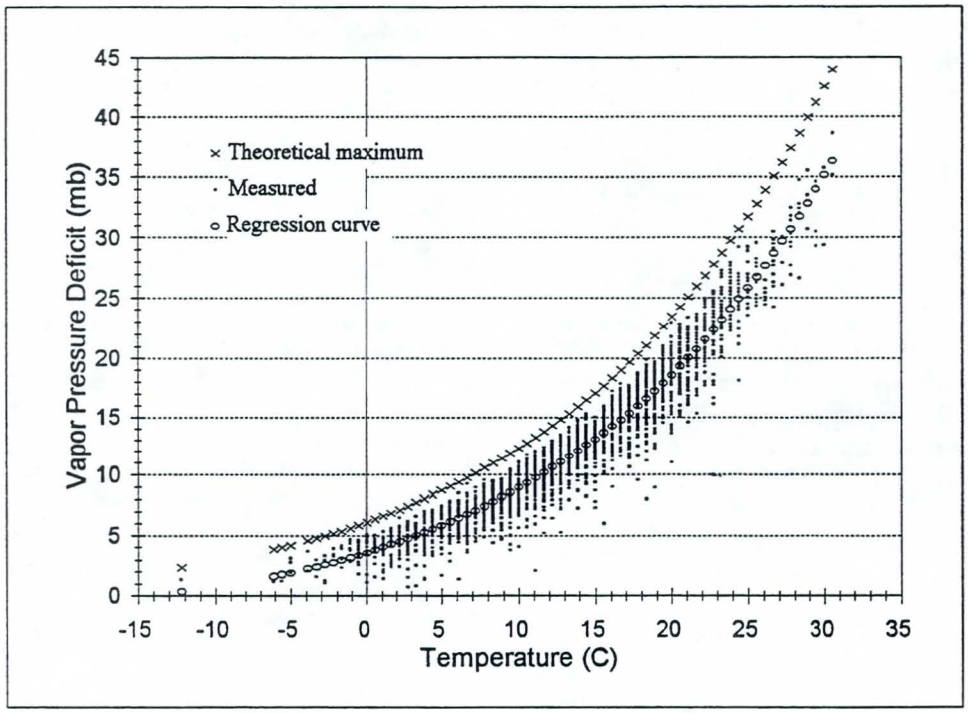

Figure 5. Vapor pressure deficit vs. temperature for days with no precipitation.

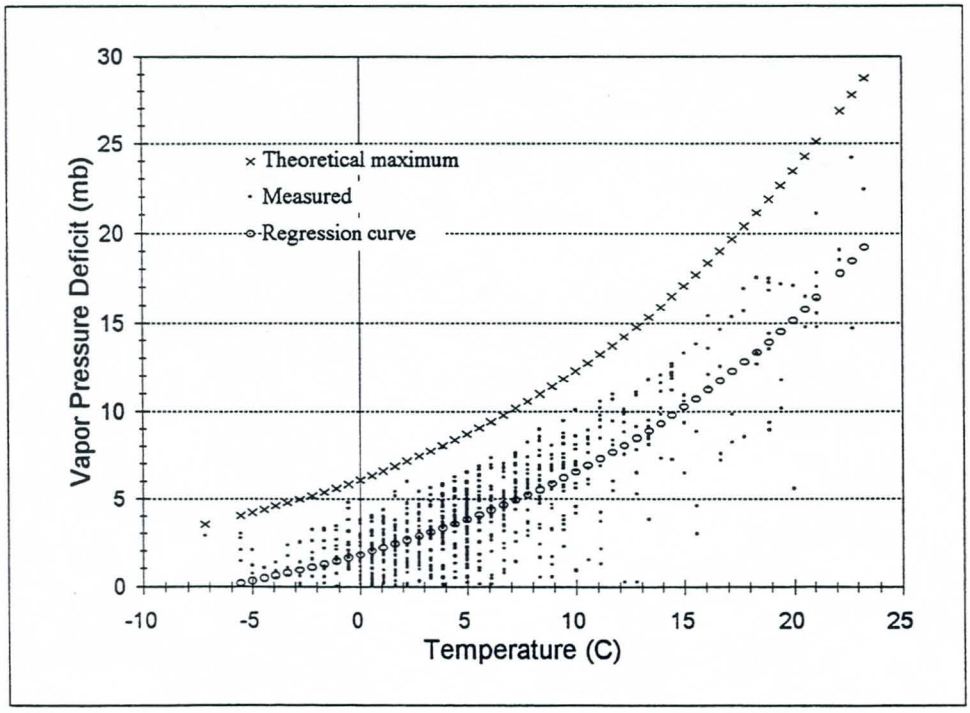

Figure 6. Vapor pressure deficit vs. temperature for days with precipitation.

random number generator to produce values for deviation from the regression curve, assuming a normal distribution for the error of the regression. In addition, the tendency for temperature to show persistence from one day to the next is described using a lag-one Markovian process.

An examination of the humidity data shows that for days without rain or snow, the vapor pressure deficit can be described simply as a function of temperature. The regression curve of vapor pressure deficit versus temperature has an R^2 value of 0.93, indicating that the regression equation is accounting for most of the variability in the vapor pressure deficit. During wet days of precipitation, the vapor pressure deficit can also be described as a function of temperature, but the relationship is not as strong. In such cases, the value of R^2 is 0.74 and the vapor pressure deficit shows a greater variability than on dry days.

In conclusion, the stochastic model described in this study successfully simulates the daily temperature and humidity patterns throughout the cold season at a site in the ponderosa pine forests of north-central Arizona.

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REAUTHORIZATION OF THE SAFE DRINKING WATER ACT AND THE VARIABILITY OF RURAL PUBLIC WATER SYSTEMS

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The provision of safe, high-quality drinking water at affordable prices is crucial in maintaining the quality of life in all communities. Industrial, commercial, residential, and recreational uses of land require a dependable supply of drinking water to be viable. This resource dependency is particularly dramatic in rural areas since many communities not only seek to maintain the existing quality of life, but frequently pursue economic growth as a means towards higher incomes and full employment.

The ability of rural communities to provide safe and affordable drinking water is determined in part by the existing quality of supplies. As ground and surface water become contaminated, both health risks and treatment costs rise. To reduce the risks posed by water pollution, a complex legislative framework has been developed in Arizona to promote prevention, to regulate potentially polluting activities, and to require remediation when water supplies have been negligently contaminated (Pima Association of Governments Report 1994). As a result, the implementation and enforcement of surface water and aquifer protection legislation have become principal determinants of the quality of water that rural water systems have at their disposal. As pollution risks are efficiently controlled, the integrity of water supplies is bolstered, and the viability of rural water systems is enhanced.

Given the quality of water supplies, the ability of rural communities to provide safe and affordable drinking water is then determined by cost-effective treatment and delivery to consumers. Treatment and delivery, in turn, are largely the concern of public water systems, defined as systems that serve piped water to at least 15 service connections or regularly serve an average of at least 25 people each day at least 60 days per year (United States Environmental Protection Agency 1993). The Safe Drinking Water Act (SDWA) and its amendments regulate public water systems. Under the SDWA, rules have been promulgated that establish drinking water standards for contaminants, treatment techniques, sampling regimens, record-keeping procedures, and public notification protocols when SDWA requirements have been violated.

The net impact of implementing and enforcing the SDWA has been to simultaneously decrease health risks posed by drinking contaminated water while dramatically increasing the cost of treatment and delivery. Compliance costs have been particularly burdensome for small systems. The Environmental Protection Agency (EPA) has estimated that compliance across all systems will cost $1.4 billion annually with many systems having to install new equipment. The adverse impact of these costs on small-system viability is evidenced by the fact that 70 percent of recent SDWA violations have been by small systems (GAO Report) and by forecasts that new requirements will exacerbate compliance problems for these systems still further.

In September 1993, the EPA submitted to Congress its "Administration Recommendations for Safe Drinking Water Act Reauthorization," EPA Office of Ground Water and Drinking Water, including ten major recommendations for revisions to the SDWA. Action on the recommendations was deferred to the 104th Congress. For rural communities, it is clear that congressional decisions on reauthorization will dramatically impact not only the current viability of small public water systems but also the future plausibility of establishing new systems as needed.

In the following sections, the SDWA is described and the SDWA Reauthorization is discussed, followed by an identification of issues and needs posed by its potential implementa-

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tion, particularly as it applies to rural areas dependent upon small public water systems.

Overview of the Safe Drinking Water Act

The Safe Drinking Water Act (SDWA) was enacted by Congress in 1974 and was amended as recently as 1986. The purpose of the Act is to ensure that drinking water supplied to the public is safe—that is, free from contaminants that could adversely affect human health. It is the EPA's responsibility to promulgate regulations that carry out the provisions of the Act. In particular, the EPA is required to set standards and to identify treatment techniques for contaminants, and to establish requirements for monitoring water quality and for ensuring the proper operation and maintenance of water systems (Fennemore Craig). Water suppliers are responsible for making sure that the water meets EPA standards and for complying with established monitoring, operation, and maintenance protocols. However, it is important to note that the SDWA does not provide funding to support mandated treatment activities.

In Arizona, primary enforcement responsibility for the SDWA is with the state through the Department of Environmental Quality (ADEQ). The EPA plays an oversight role providing guidance, technical assistance, and some financing. While the state has been delegated "primacy," actual enforcement relies heavily upon community water systems demonstrating compliance through periodic sampling and testing requirements. In the unlikely event that state enforcement is inadequate, emergency federal enforcement provisions are available to the EPA in the form of issuing orders for public notification of SDWA violations, mandating clean-up, requiring the use of an alternative supply, and/or imposing daily fines.

There are three major types of requirements in the SDWA: (a) sampling and reporting, (b) record keeping, and (c) public notification (U.S. EPA 1993). Each supplier of water must collect samples from the water system, take them to a certified laboratory for analysis, and send the results to DEQ. The laboratory results, name of the person who collected the samples, dates and locations of sampling points, steps taken to correct problems, sanitary survey reports, and other information must be kept on file by the water supplier. Finally, any time there is a violation of a requirement, the public must be notified.

One significant advantage to the state assuming enforcement primacy is that some degree of flexibility can be exercised by the DEQ in implementing the Act. For example, some requirements can be made stricter, such as requiring operator certification or minimum design standards. On the other hand, variances and exemptions can be issued from some of the requirements for systems that are having major technical or financial problems associated with compliance.

There are two types of drinking water standards that apply to all public water systems in Arizona: primary and secondary. Primary standards are health based and enforceable. Secondary standards are based on the aesthetic quality of the water and are non-enforceable guidelines. In the case of primary standards, maximum contaminant levels (MCLs) are concentrations that are judged to be associated with acceptable health risks given cost and technology constraints. For chemicals that are believed to cause cancer, the goal is to set MCLs as close to zero as is technically and economically feasible. For contaminants that are difficult or costly to measure, treatment techniques are required in lieu of specifying an MCL. Secondary MCLs have been established as guidelines associated with the aesthetic quality of water, such as taste, odor, or color, and are not enforceable.

Underlying any discussion or evaluation of the SDWA is an inescapable tension between capturing the documented health benefits associated with drinking safe water and bearing the significant costs of precaution. Monitoring turbidity, bacteria, total chloroform, lead, copper, and radionuclides, as well as inorganic, synthetic organic, volatile organic, and chlorinated organic chemicals, can help reduce a wide array of health risks, varying from gastroenteric infections, to liver and kidney damage, to several types of cancer. Unfortunately, the additional costs imposed by monitoring, sampling, treatment, and record keeping are substantial, and for many small water systems they are particularly burdensome.

The Eastern Pima County SDWA Experience

The legislative intent of the SDWA is to ensure that drinking water poses minimal risks to public health. The EPA is charged with implementing the intent of the Act and does so in two steps. First, "safe" drinking water is defined

during the process of setting MCL standards and establishing treatment techniques. That is, drinking water that is in compliance with MCL standards and treatment requirements is judged to be safe for public health purposes. Second, protocols for sampling, record keeping, and public notification have been established to promote the compliance of public water systems with EPA treatment and MCL safety standards.

In general, sampling requirements are detailed and complex, but the overall intent is to tailor sampling procedures to the type of containment being analyzed. The requirements address the types of analyses to be performed, the frequency of sampling, the location within the water system where sampling must occur, preservation techniques, transportation precautions, and laboratory certification. Some public water systems (e.g. small water systems, such as systems serving less than 3,300 people, or transient, non-community systems, such as systems serving hotels or restaurants) may receive variances or exemptions to these requirements.

In addition to sampling activities, public water systems are required to keep records on several aspects of sampling, including chemical analyses, MCL violations, enforcement actions, and sanitary surveys. Upon analysis of a constituent, a certified laboratory must report the results to ADEQ within 3 working days. If the analysis shows an MCL violation, then reporting must be within 24 hours of the completion of the analysis. All systems, regardless of distinguishing factors, are responsible for reporting all chemical analysis results, violations, and public notices to ADEQ.

In the event of a violation, public notification takes numerous forms, such as hand delivery, electronic media, continuous posting, and direct mail, in order to ensure that affected individuals will be adequately informed. Notification procedures are described for each type of compliance violation. Two categories of violations are distinguished: Tiers 1 and 2. Tier 1 violations pose serious and direct risks to human health through either chronic (non-acute) or brief (acute) exposures. Exceeding an MCL or violating treatment technique requirements are examples of Tier 1 violations. Tier 2 violations are less directly threatening to human health and generally violate SDWA specifications on a procedural basis. Examples include a failure to monitor the water supply or to follow prescribed sampling and analysis methods. In general, public notices must include a discussion of the violation, the potential for adverse effects, the population at risk such as children or pregnant women, the steps taken to correct the problem, and recommended precautions.

ADEQ and the State's Primacy Role

The drinking water program was designed so that the day-to-day responsibility to carry out the program would be delegated to approved state governmental agencies while the EPA provided guidance, assistance, and limited funding. The state of Arizona has been granted primacy through ADEQ and therefore accepts the obligation to monitor and enforce EPA requirements pertaining to SDWA. To assess compliance, ADEQ has established rules and procedures that address the production, treatment, distribution, and testing of public water systems.

The determination of compliance unfolds through an inventory and analysis of each system. A key component is the sanitary survey, an on-site review of the system's water source, facilities, equipment, operation, and maintenance, performed approximately every 5 years. Upon inspection, a compliance status of 1, 2, or 3 is determined. A system operating under full compliance with the SDWA will earn a rating of one.

A compliance status of two indicates that a system is in general compliance with the act's provisions and is considered to pose only moderate risks to public health. Operational and maintenance procedures are most often the cause of violations; however, sampling procedures are also potential candidates. Examples of level two violations include inadequate site clean-up, lack of proper fencing and security, susceptibility of a system to freezing, or user complaints. The issuance of an administrative order is the generally accepted practice for redressing "substantial-compliance" violations.

A compliance designation of three denotes a system in non-compliance. Systems in non-compliance may have exceeded an MCL, failed to properly implement treatment requirements, ignored operation and maintenance procedures, or simply failed to sample the water. Examples of an operational and maintenance violation that would result in a rating of three include failing to install a pressure gauge, failing to chlorinate when necessary, or not having the well site graded properly. Initially, correction of a violation is addressed through an administrative

order. If the system fails to comply, the order may proceed to full closure of the system.

SDWA Compliance in Eastern Pima County

Recent compliance of small public water systems with respect to the SDWA was analyzed. Data was collected from ADEQ that reflects the compliance status of public water systems as recorded in their most recent system survey prior to 1994. Not reflected in the analysis are small systems dependent upon surface water. In narrowing the focus to groundwater, only six systems in Pima County were eliminated. Similarly, compliance of large public water systems (e.g. Tucson Water) is not recorded.

Only 18 of the 238 systems evaluated were awarded full compliance with the SDWA, while substantial compliance was granted to an additional 98 systems. The remaining 165 systems were classified as being in non-compliance with the SDWA.

The probability of a public water system being in compliance or substantial compliance with SDWA regulations does not appear to vary systematically with system size. ADEQ (1994) defines systems of medium size as serving between 3,301 and 50,000 individuals, with small systems serving less than 3,300. In this survey of recent compliance performance, 46 percent of the medium systems were discovered to be in non-compliance while 51.5 percent of the small systems were classified similarly.

Approximately 95 percent of the public water systems in eastern Pima County are classified as small. Of the small systems, 49.5 percent were in compliance or substantial compliance with the Act. In 5 cases, comprising slightly more than 2 percent of the small systems, water quality failed to meet MCL standards, thus potentially posing serious risks to human health. The remaining 112 non-compliant systems were in violation of SDWA regulations in ways unrelated to MCL and treatment technique requirements. Specifically, the systems failed to comply through improper operator certification (64.3%), inappropriate sampling procedures (57.1%), unsuitable operation and maintenance (17.8%), or some combination of the above (34.8%). For these systems, failure to fully comply with the SDWA may not reflect increased health risks as much as an inability or unwillingness to bear the financial burdens associated with EPA protocols.

SDWA Cost Implications for Small Public Water Systems

The 1986 amendments to the SDWA mandated a dramatic increase in the number of drinking water contaminants to be regulated. The potential economic impact of this regulatory expansion was recently assessed by ADEQ (1994). For small systems, ADEQ expects that investment costs could be substantial when treatment becomes necessary. For example, the capital cost of treating inorganic contaminants is estimated to vary between $61,000 and $135,000 per system. While these potential costs are high, the likelihood that they would actually be incurred is low since groundwater quality is good to excellent over most of the Tucson Basin (PAG draft report 1994).

While additional capital costs may be avoided by small systems in Pima County, the same cannot be said of sampling costs. The 1986 amendments called for regulating an additional 66 contaminants by 1989 and for further expanding the number of MCLs by 25 every year thereafter. The sampling costs associated with this regulatory expansion are likely to become progressively burdensome if small systems are to comply.

Recent evidence documents that a significant number of small public water systems are already in non-compliance with the SDWA. As new regulations are promulgated and enforced, systems can choose to comply partially, merge with other systems, borrow required investment funds, charge higher prices to consumers, some combination of the above, or shut down. The relative desirability of these alternatives will be greatly affected by the specification of the SDWA reauthorization bill currently being debated in Congress.

Reauthorization of the Safe Drinking Water Act

In the 1993 version of the reauthorization, the EPA submitted to Congress an extensive list of proposed revisions to the SDWA (Trager et al. 1994). The original reauthorization bill, sponsored by Senator Max Baucas, included many of the recommendations proposed by the EPA, but failed to gain bipartisan support due to a failure to address the MCL standard-setting process. A second bill was subsequently introduced requiring increased risk/benefit analysis in standard setting, but was opposed by the EPA on the

grounds that the mandated analysis would unrealistically delay the issuance of regulations. Recently a flurry of bills and amendments have been introduced designed to address EPA and public water system concerns, each enjoying some degree of support. Ultimately, successful legislation will have to address a variety of concerns. Particularly prominent among these concerns is the task of safeguarding public health while limiting the financial burden imposed by unfunded federal mandates.

Several of the EPA's recommendations have general applicability across public water systems regardless of size. For example, the EPA wants to maintain and strengthen state primacy, a position strongly supported by ADEQ. In addition the recommendations call for mandatory state programs to protect ground water and surface water supplies, and for mandatory minimum operator certification program criteria to apply to all water systems, including small systems. Programs to protect water supplies have been in existence for many years in Arizona, and have been substantially revised and updated since 1986. Similarly, operator certification programs are already operational. In these regards, Arizona has acted proactively in an attempt to strengthen the state's ability to provide affordable, safe drinking water.

Of direct concern to small public water systems are the following EPA recommendations:

- Establish (and adequately funding) a Drinking Water State Revolving Fund to provide low-interest loans to help water systems meet the costs of SDWA compliance.
- Require state-implemented programs to assess the viability of existing small systems and to prevent the formation of new, nonviable systems while restructuring and/or consolidating nonviable, noncompliant small systems.
- Establish a less expensive "best available technology" that small systems could use to comply with the SDWA, if they would not otherwise be able to achieve compliance through restructuring.
- Allow for longer compliance deadlines for making drinking water standards effective, moving from 18 months after EPA promulgation to 60 months.

Noticeably absent from the list of EPA recommendations are suggestions for revising the standard-setting process itself. Between 1974 and 1986, the EPA issued rules regulating 23 drinking water contaminants. The 1986 SDWA amendments required the EPA to establish national drinking water standards or treatment techniques for 83 contaminants by 1989, and for 25 additional contaminants every 3 years thereafter. Critics of the program argued that this regulatory expansion places a disproportionate burden on small public water systems in rural areas by failing to account for local health-risk impacts and budgetary realities. That is, failure to base drinking water standards on a site-specific, risk-benefit basis runs the risk of burdening drinking water programs to the point of collapse.

Future Outlook

Supporters of strict SDWA regulations are quick to argue that evisceration of the program is likely to be "penny wise, but pound foolish," since the high price of water treatment is more than justified by the cost savings associated with the prevention of disease. For example, the EPA estimates that the Surface Water Treatment Rule alone helps avoid 90,000 cases annually of acute gastroenteritis, and that the Lead and Copper Rule can reasonably be expected to reduce exposure to 140 million people, including 18 million children, to unsafe levels of lead in their blood (GAO Report).

Unfortunately, many small systems do not comply with SDWA standards or monitoring requirements due to disproportionate costs. Similarly, according to the EPA, the primary reason for state drinking water non-compliance is resource scarcity, as reflected in prohibitive costs to implement new regulations, competition with other state programs for scarce financial resources, legislative priorities, and state budget shortfalls.

In recognition of both health and cost concerns, Arizona will be participating in an EPA national survey of drinking water systems this fall (Water Resources Research Center 1994). The objective of the study is to determine the level of investment needed in the nation's drinking water systems to supply safe water and comply with current and future federal regulations. Assisted by ADEQ, the EPA will survey public water systems across the country, both large and small, to develop computer models to project capital investment needs of drinking water systems.

There appears to be consensus in the regu-

lated community that an appraisal of investment needs is a crucial first step in building a workable program to safeguard public drinking water in light of budgetary realities. From the EPA's perspective, the survey initiative is part of a general redirection effort with respect to the safe drinking water program (McGuire 1995). A comprehensive plan (announced in late December, 1994) is being developed over the next 5 to 8 months in which the agency intends to emphasize shifting resources away from low risk to high risk regulations, and expand the use of regulatory negotiation among affected parties in implementing costly treatment rules. While a great deal of uncertainty remains about the net impact of the water system survey, the EPA's redirection program, and the reauthorization debates in Congress, one effect on consumers of drinking water in rural areas is inescapable: As consumers demand higher quality water, and stricter drinking water standards are implemented, public water systems will be forced to charge higher prices that reflect the full real cost of supply.

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