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INTRODUCTION

The Spring meeting of the Hydrology Section of the Arizona-Nevada Academy of Science took place in Las Vegas, Nevada on April 14, 2001. We want to extend our thanks to Ms. Louella Holter, Bilby Research Center, Northern Arizona University, for her editorial assistance on this publication.

Malchus B. Baker, Jr.

AN ANALYSIS OF THE PROPOSED DECOMMISSIONING OF THE FOSSIL CREEK DAM, NEAR STRAWBERRY, ARIZONA

Charles E. Jones Jr. and Patrick Phillips*

In 1916, the Fossil Creek dam was built near Strawberry, Arizona to provide power for rural communities throughout the Verde Valley. Before 1916, Fossil Creek was fed by springs at a rate of 3.75 million liters per hour. One of Arizona's most productive and diverse ecosystems, this perennial stream served as a unique riparian habitat that supported native fish and a diverse assemblage of native Arizona flora and fauna. However, the hydropower operation diverted nearly 100 percent of the flows from Fossil Creek, leaving 22.4 km of the stream channel dry, ecologically degraded, and with little aesthetic value. In an effort to restore the creek, a coalition of environmental organizations has recently signed an agreement with Arizona Public Service to decommission the dam by 2005. The decommissioning will include partial removal of the dam and other related structures, leading to a complete restoration of the ecosystem by 2009. This analysis evaluates the alternatives for decommissioning the Fossil Creek dam to restore the stream to its proper functioning condition. Removing the dam and returning the full flows will speed the restoration process, but two areas that remain of concern are the proliferation of exotic fish and vegetation into the restored stream channel and post-restoration recreational impacts. A management plan should be developed prior to the initiation of restoration activities to protect against these impacts.

Introduction

Fossil Creek is located at the southern limit of the Colorado Plateau, in north-central Arizona, just below the edge of the Mogollon Rim. The creek and its associated canyon lie in an isolated region just northwest of the town of Strawberry (Figure 1). Historically one of Arizona's most lush and beautiful productive and diverse ecosystems, the watershed consists of a perennial stream flowing 22.4 km from a system of springs (Fossil Springs) to the confluence with the Verde River.

The series of seven springs produces a constant combined flow of 1218 L/s (Malusa 1997). This usually accounts for the full flow volume of the drainage, while seasonal run-off adds to the flow regime 20 percent of the time (Loomis 1994). The water released at Fossil Springs contains high concentrations of calcium carbonate ($CaCO_3$), resulting from groundwater flowing through a limestone geological formation. When groundwater saturated with $CaCO_3$ surfaces, $CaCO_3$ precipitates in the form of arc-shaped travertine dams. Over time, this process had created a stunning system of pools, riffles, and waterfalls in a unique riparian area nestled among the deserts of Arizona. Above each dam lay the clear blue pools that are characteristic of travertine-forming waters. Water cascaded from one pool into the next, oxygenating the waters for the fish that had been trapped within the pools. This unique environment was an ideal "natural fish hatchery" that supported a wide variety of native Arizona fish and fauna (Mockler 1999).

In the southwestern United States, water is a treasured commodity, especially for the flora and fauna that rely upon it. Riparian vegetation thrived at the fringes of the pools in the perennial waters of Fossil Creek. Riparian areas are very important habitat for birds, insects, reptiles, amphibians, and fish, and the restoration of these essential ecosystems has become a national priority (National Research Council 1992).

Historical Background

Historically, each travertine dam within the creek corridor grew approximately 1.0 m^3 every 43 days (Malusa 1997). Flash floods are a common occurrence of lower order streams in the southwestern United States; they accumulate intensity quickly and dissipate rapidly. Floods frequently destroyed or displaced the large travertine depos-

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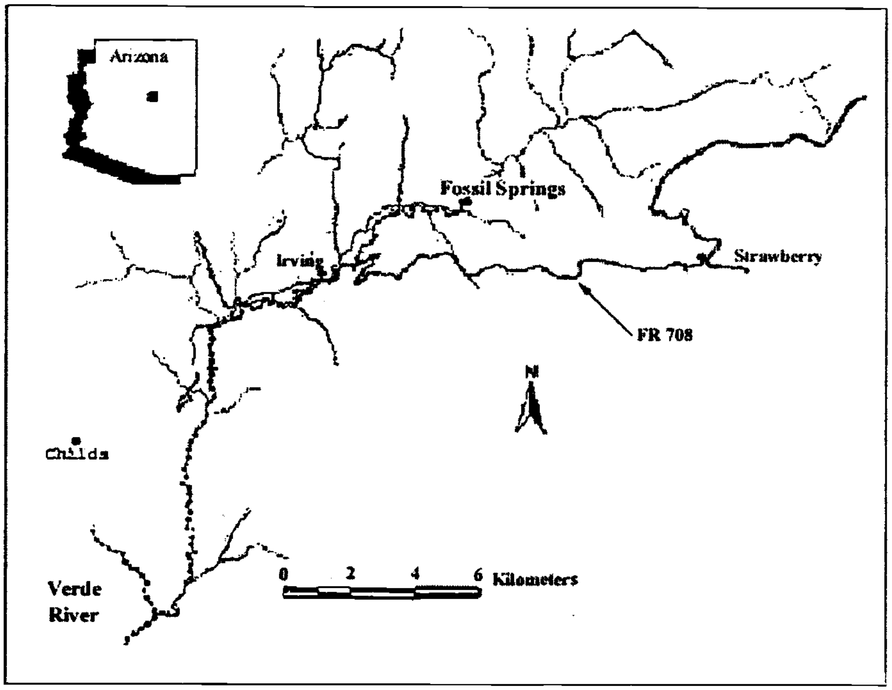

Figure 1. Fossil Creek and surrounding area. Map courtesy of USFS Rocky Mountain Research Station.

its, but new dams and pools were reestablished relatively quickly. After the initial 6.4 km below the springs, the CO_2 concentrations in the water equilibrated with the atmosphere and travertine deposition ceased.

Sycamore appeared to dominate the ecosystem overstory, and aquatic grasses were prevalent along the stream banks (Sayers 1998). The remaining 17 km of riparian corridor were lush and heavily utilized by the native species of the area. In the early 1900s, the perennial flow from Fossil Creek springs was tapped to produce hydropower for the growing communities in the Verde Valley. The Fossil Creek dam was constructed in 1916 (Figure 2), 250 m downstream from the springs; thus began the diversion of the 1218 L/s flow provided by Fossil Springs. The carbonate-rich water is now rerouted through a 6.7 km long flume to a hydroelectric power plant located at Irving, Arizona. At the Irving power plant, approximately 5.6 L/s of water escapes back to the streambed as normal seepage. The remaining flow continues another 7 km to the Childs power plant (Malusa 1997). The highly efficient operation (Figure 3) generates 5.6 megawatts per year (American Rivers 2001), which represents less than 0.1 percent of the total annual power production by Arizona Public Service (APS). It is worth less than $585,000 annually (Malusa 1997).

In 1994, APS applied to the Federal Energy Regulatory Commission (FERC) to renew their license for water diversion from Fossil Creek for power generation. As part of the re-licensing process APS was required to submit an environmental assessment to FERC. Upon review of this document many questions were raised regarding the unique qualities of the Fossil Creek watershed. A coalition of conservation and environmental organizations took an active interest in the re-licensing process and eventually negotiated a legal agreement with APS to decommission the Fossil Creek dam and restore full water flows to Fossil Creek by 2005 (American Rivers 2001). Furthermore, APS has agreed to remove the top 2 meters of the dam, including the intake structure and the entire aboveground flume system, and to restore the maintenance road to a hiking trail, by the year 2009.

Environmental Impacts of the Dam

No environmental assessments were completed prior to the construction of the Fossil Creek diversion structure. Therefore there are no data to compare pre- and post-dam conditions. To learn

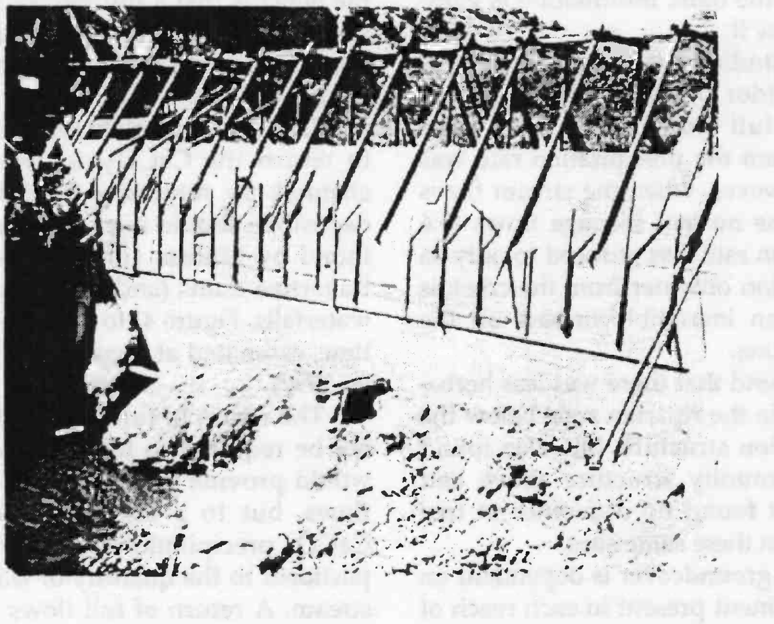

Figure 2. Construction of the Fossil Creek dam started in 1916. Photo courtesy of Arizona Public Service.

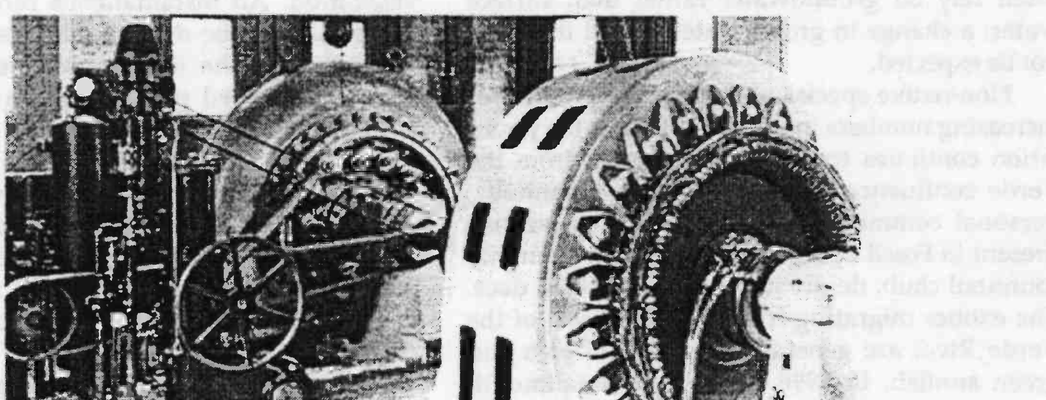

Figure 3. The simplicity of the Irving power plant (1916). Photo courtesy of Arizona Public Service.

about the effects of the dam, information is gathered above and below it.

Malusa (1997) studied the travertine deposition along the corridor of Fossil Creek. Malusa found that when full flows (1218 L/s) were restored to the stream the precipitation rate was 11,952 kg/day. However, when the stream flows were returned to the normal seepage flows (5.6 L/s) the precipitation rate was reduced to only 46 kg/day. The diversion of water from the creek is obviously having an incredible impact on the deposition of travertine.

Sayers (1998) found that there was less herbaceous groundcover in the riparian zone below the Fossil Creek diversion structure. She also found differences in community structure above and below the dam, but found no difference in tree growth rates between these same sites.

The herbaceous groundcover is dependent on the amounts of sediment present in each reach of the stream. The reduction in water flows also prohibits the transport and subsequent deposition of sediment in the system. The differing community structures are artifacts of differences in the amounts of surface water, stream morphology, canyon geomorphology, and travertine deposition, among other factors (Sayers 1998). Riparian trees often rely on groundwater rather than surface water; a change in growth rates would therefore not be expected.

Non-native species of fish are present in ever increasing numbers in Fossil Creek and the population continues to establish upstream from the Verde confluence to the Irving dam (Sponholtz, personal communication 2001). The native fish present in Fossil Creek below Irving are primarily roundtail chub, desert sucker, and speckled dace. The exotics migrating from the main stem of the Verde River are generally smallmouth bass and green sunfish. In 1996, 2–3 pound smallmouth bass were observed just below the Irving power plant. Razorback suckers have been introduced above the diversion dam 6.4 km upstream, but sampling has found no evidence of this species persisting in the area.

Restoration Alternatives

The effect that hydroelectric power generation is having on Fossil Creek seems apparent. An obvious next step in the research process would be to look at ways of restoring the stream. Ideally, a restoration analysis of Fossil Creek should examine several things: the costs of maintaining the status quo, the costs of decommissioning the dam, the benefits that a full restoration would have for the flora and fauna, the potential recreational impacts to result from the restoration, and how to minimize those impacts.

The most important aspect of the restoration is to return the $CaCO_3$-rich waters to the stream channel. By returning the full flows, travertine deposition should increase to rates similar to those found by Malusa (1997). These rates will allow travertine dams (and the characteristic pools and waterfalls, Figure 4) to rebuild in a relatively short time, estimated at a maximum of 10 years (Mockler 1999).

The return of full flows to Fossil Creek would not be required to fulfill this need. Partial flows would provide the ecosystem services of the full flows, but to a lesser degree. The amount of $CaCO_3$ precipitation to be achieved will be proportional to the quantity of water returned to the stream. A return of full flows would be ideal for ecosystem restoration, but there are some parties who argue against the restoration of any flows. The return of partial flows would represent a compromise option between the two viewpoints.

A return to the natural stream morphology will allow for increased sediment deposition, providing substrate for the establishment of riparian vegetation. An instantaneous return of the full flows may not be desirable for fish and riparian vegetation in the long term. These inhabitants have experienced periodic high flows and have sustained their populations, but a gradual return of full flows would be most desirable. Ramping the flows may allow the riparian vegetation to migrate up the stream bank without being instantaneously flooded and destroyed. If a mass killing of riparian vegetation does occur, the public may desire plantings of vegetation along the stream channel.

Among experts there is some disagreement over whether a return of flows may allow invasive fish species to infiltrate the system by allowing them to swim up the stream channel. W. L. Minckley has listed three species of fish that presently survive above the diversion dam, but believes that Fossil Creek offers a potential recovery area for the endangered razorback sucker and Gila topminnow (Mockler 1999). Minckley is convinced that the natural travertine dams will act as a barrier to invasive non-native fish (Mockler 1999).

Offering a different viewpoint, Sponholtz has suggested that "if management does not intercede and chemically renovate to remove the nonnative fishes and construct downstream barriers to fish

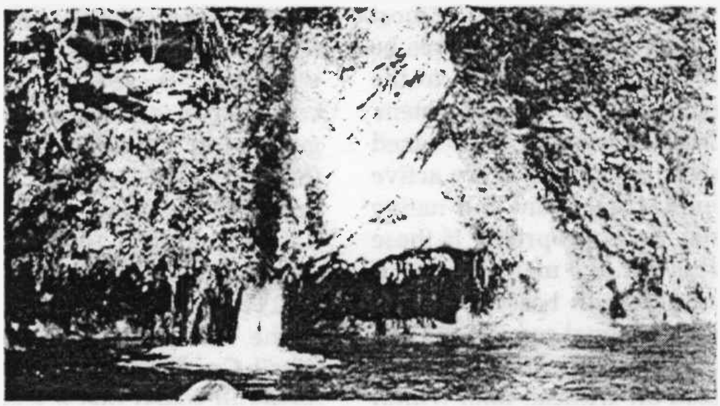

Figure 4. Travertine deposition at Fossil Springs (1916). Photo courtesy of Arizona Public Service.

movement, the outlook for the natives is grim" (Sponholtz, personal communication 2001). This course of action would likely include relocating native fish species below the Irving power plant to locations upstream of the renovation point.

Another management alternative suggested by Sponholtz (personal communication 2001) would entail promoting Fossil Creek as a roundtail and smallmouth fishery and removing all bag limits in the hopes that sport-fishing take would mitigate the impact of invasive smallmouth predation on endangered indigenous fish. Increased visitation as a consequence of re-watering would likely result in exotic fish introduction by fishermen, making this idea attractive. The management of flows to discourage non-native fecundity is an alternative that deserves consideration, but would require further study.

APS has agreed to remove the top 2 meters of the dam. In terms of restoration, a partial removal of the dam will not be necessary. The dam will be quickly covered with travertine and will soon resemble a large travertine waterfall. The United States Forest Service (USFS) is calling for APS to remove the entire structure. Their position is that the dam should be removed while it remains under APS liability; otherwise if the structure failed, expensive clean-up efforts would be required by the USFS.

The restoration of Fossil Creek will result in a unique and lush ecosystem. The recreational impacts to such a system could be tremendous without proper planning. Some ideas have been discussed, but there are no plans for such accommodations. There are many techniques to minimize recreation impacts, including developing park infrastructure through trails, picnic areas, restroom facilities, and camping sites. These areas would offer protection through sacrifice of "non-essential" areas along the stream corridor. Pam Sponholtz (personal communication 2001) stated that the USFS has asked APS to leave the structures near the Irving facility. These could be used as cabins for visitors or administrative buildings for the USFS. Cabins might provide some income for the USFS to maintain the area. Sponholtz also remarked that there was some discussion regarding the closing of FR 708, which leads from a trailhead at the top of the watershed to the Irving power plant and beyond. This could cut down the number of visitors because it would require a 3.2 km hike to the water. This seems like a good idea, with special permits being offered to handicapped persons, researchers, and others with special needs to allow easier access to the area.

Restoration Recommendations

A passive restoration approach seems appropriate for Fossil Creek. From an ecological perspective, the return of the full 1218 L/s to the stream channel will be ideal. This will allow for the characteristic travertine dams to develop and offer a structurally diverse channel. Increased sedimentation would enhance natural recolonization by riparian vegetation. To facilitate this re-colonization, the flows should be ramped to full flows over a period of 6 months to a year. This ramping period should consider the natural reproduction cycle of the plants and try to facilitate the re-colonization process.

The alternatives for the fish are difficult to assess. It is therefore recommended that the most conservative and therefore cheapest measures be undertaken first. This would be to return to the full flows, allowing fish to migrate into the system. If the higher Fossil Creek ecosystem is colonized by invasive exotic fish species, then more active measures such as chemical treatment and native fish reintroduction may be appropriate. If these efforts are unsuccessful, then the management of the ecosystem as a smallmouth bass fishery as suggested by Sponholtz (personal communication 2001) may become a reasonable alternative.

The dam has created an artificial pool within higher Fossil Creek. These pooled waters have allowed for the invasion of exotic vegetation that prefers slow-flowing water, thus serving as a source population to possibly invade the lower stretches of Fossil Creek. The calm waters could support populations of fish and invertebrates that would not be able to exist in the natural flowing waters of Fossil Creek, but there is no evidence of non-native fish existing above the dam (Sponholtz 2001). For these reasons, reasonable measures should be taken to remove the exotic vegetation prior to the removal of the dam and more research should be done on the invertebrate and fish populations within Fossil Creek.

Because of the biological impacts of the dam, the upper part of the dam needs to be removed (as agreed upon in the signed agreement). Although the remaining dam structure would probably not be a threat to the restoration process, the USFS must consider the liability issue. Risks of structural failure increase with age; therefore APS should be required to completely remove the structure.

Finally, measures must be taken to anticipate the increase in recreational traffic. These measures should include designated parking areas, trails, picnic areas, restroom facilities, and camping sites. By offering facilities, the USFS will be able to direct the recreational impacts to specific areas. Closing FR 708 to the general public could prove highly desirable. This could be used as a USFS access road and permits could be offered to research groups and citizens with special needs. These measures would minimize recreational impacts and increase the chances of a successful stream restoration.

Alternatives to Restoration

There are alternatives to complete restoration. People pushing for a less than complete restoration include Sam Steiger (mayor of Prescott, Arizona), James Doolittle (Flagstaff consultant), and Dan Israel (Gila County Consultant). Steiger has written to the FERC stating that the Prescott community is interested in obtaining the power generating facility for their growing population (Steiger 2000). Doolittle has been searching for a group to purchase the water rights and FERC license from APS. If successful, he stands to gain a substantial profit from the transaction (Sponholtz 2001). Israel recently approached Jerome Stefferud of the USFS about using the diverted waters of Fossil Creek in Gila County for residential use. Stefferud stated that it was not feasible, but Israel was not curtailed (Stefferud 2001).

Acknowledgments

We'd like to offer special thanks to Pam Sponholtz (AZ Department of Game and Fish), Mindy Schlimgen-Wilson (Nature Conservancy), Phil Smithers (APS), Rod Parnell (NAU Geology), Matthew Loeser, and Bradner McRae. Each was a great help in various aspects of the analysis and in discussing the Fossil Creek dam and its decommissioning.

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MANAGEMENT OF ARID AND SEMI-ARID WATERSHEDS: DECISION-MAKING TOOLS AND TECHNOLOGY TRANSFER

Malchus B. Baker, Jr.,[1] Barbara Hutchinson,[2] Eric Pfirman,[2] Michael Haseltine,[2] and Jeffrey Schalau[3]

Abstract

The Central Arizona Highlands have been the focus of a wide range of research efforts to learn more about the effects of natural and human induced disturbances on the functioning, processes, and components of the ecosystems found in the region. The use of current electronic technologies provides a unique reference and educational tool for disseminating research findings to scientists, educators, and land management professionals as well as the general public. Phase 1 of this project provided a variety of background material about watershed management, and specifically the Beaver Creek Experimental Watershed, which is accessible through the World Wide Web. This report discusses how 20 years of data collected on the various experimental watersheds are being put into a Web-based relational database and geographic information system (GIS) that will allow the data to be searched and graphed. The database incorporates a variety of factors regarding precipitation, streamflow, sediment, water quality, vegetation, and wildlife. Additional data from chaparral shrubland are being added that contrast with, and can be compared to, information from the ponderosa pine and pinyon-juniper vegetation types.

Introduction

In the late 1950s, the Beaver Creek Experimental Watershed (BCEW) project became a significant component of the Arizona Watershed Program (Fox et al. 2000). This program was initiated to evaluate the usefulness of selected vegetation management programs intended to increase water yields and other multiple resource benefits in the Salt and Verde River basins (Baker 1999). Studies were conducted in mixed conifer (Baker 1999; Rich and Thompson 1974; Rich and Gottfried 1976; Gottfried and Ffolliott 1992), ponderosa pine (Baker 1999; Baker 1984; Brown et al. 1974), pinyon-juniper (Baker 1999; Baker 1986; Clary et al. 1974), and chaparral (Baker 1999; Hibbert et al. 1974). The objectives of the Arizona Watershed Program were to provide land managers with facts about the effects of resource management on these vegetation types and to assess land management options to develop better plans and programs.

In 1997, former Arizona Watershed Program personnel from the USDA Forest Service and faculty from the University of Arizona collaborated on a project to make available to the public the research data and reports resulting from many years of work on the BCEW. This effort resulted in a successful grant proposal to the International Arid Lands Consortium (IALC) for a 3-year Phase 1 project to deliver this information via the World Wide Web as well as through other dissemination activities. Specifically, the Web site Managing Semi-Arid Watersheds (http://ag.arizona.edu/OALS/watershed/index.html) was developed (Figure 1), featuring expert information that can be applied to other semi-arid regions in the world facing similar natural resource management problems (Young and Baker 1998).

Phase 1 Project on Semi-Arid Watersheds

During the course of the Phase 1 project, the home page of the Managing Semi-Arid Watersheds Web site was developed to include extensive information to meet a variety of user needs. Introductory pages describing watersheds, watershed management techniques, and public land management issues are offered as well as in-depth information on the Central Arizona Highlands and the Beaver Creek watershed. Sections include a

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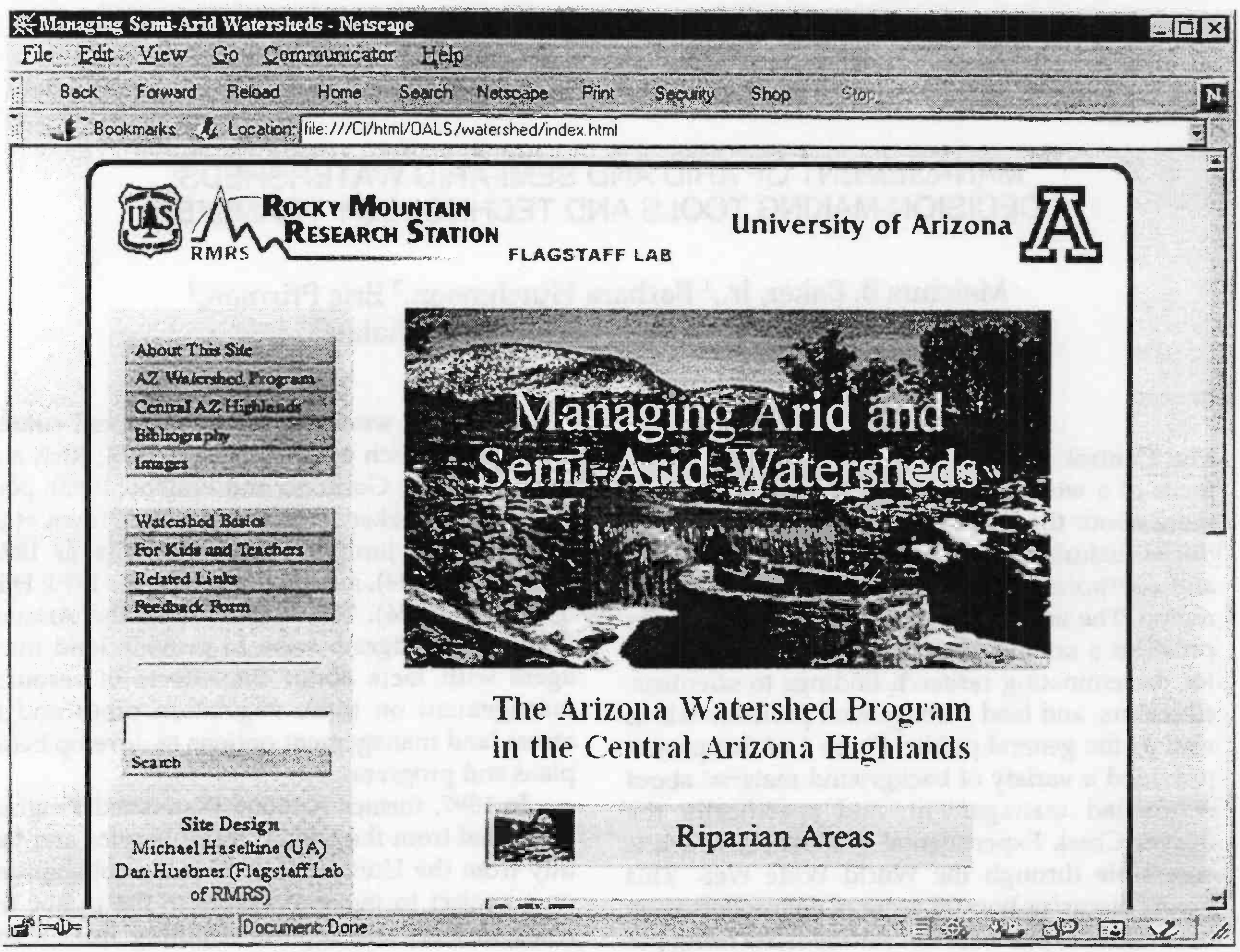

Figure 1. The Managing Arid and Semi-Arid Watersheds Home Page.

training course in watershed management developed for college students and a listing of Web resources for K–12 teachers and students. The site also features documents on selected aspects of watersheds, hypertext links to other related resources on the Web, and a search engine for the site that allows visitors to locate particular information. One of the challenges in developing this Web site was to provide enough detail to make it useful to professional researchers and land managers while still general enough for educators and students at various age levels.

To meet more advanced needs, resources that resulted directly from the Arizona Watershed Program were also made available on the Web site. These include a complete overview and description of the climate, vegetation, and watershed history of the entire Central Highlands region. A separate section, specifically on the Beaver Creek watershed, provides everything from a virtual tour of the watershed, including photographs and descriptions, to the designation of the Beaver Creek watershed as a Biosphere Reserve. However, the most extensive information on the watershed covers the treatments and management practices used over the course of the program. These summaries are accessible both through a text index and through a clickable map to identify specific watershed areas. Also included is a bibliography of 693 references to technical reports and bulletins, articles, theses and dissertations, books, and proceedings that relate to watershed research conducted on the Beaver Creek watershed or sponsored by the project.

Field days on the Beaver Creek watershed were also a feature of Phase 1 activities. These activities introduced the Beaver Creek watershed to the public, initiated planning for future educational workshops, and allowed hands-on learning for teachers, students, and the interested public.

The field days program focused on four general themes: forest management, wildlife habitat and management, rangeland management and monitoring, and watershed condition and function. Moving beyond a lecture approach, participants took part in hands-on demonstrations, conducted experiments, and performed watershed monitoring techniques. Other outreach activities included demonstrations of the Web site at technical and pubic meetings, presentations on the project at conferences, and announcements in newsletters and appropriate listservs.

Phase 2 Project on the Watersheds of the Central Arizona Highlands

In 2000, another 2-year project, called "Management of Arid and Semi-Arid Watersheds, Phase 2: Decision-Making and Technology Transfer," was funded, again by the IALC. Its primary goal is to create innovative electronic tools for watershed management, based on the experiences of and knowledge gained through the Arizona Watershed Program, coupled with a new program for educational outreach to the community.

There are five distinct but interrelated objectives in this phase of the project. The first is to create a relational database from the 20-plus years of BCEW data and a Web-based interface for easy accessibility of the data to researchers, land managers, educators, students, and the general public. In this way, the data can be used as a decision-making and teaching tool, and the database structure can be used as a template for other watershed data projects in arid and semi-arid regions around the world. The second objective is to create interactive Web-based maps for viewing the areas where the BCEW and monitoring stations were located and for selecting and viewing the data collected at those stations. The third objective is to inventory and list information and types of data collected on other watersheds in the Central Arizona Highlands for eventual inclusion on the Managing Arid and Semi-Arid Watersheds Web site. Related to this is the fourth objective which is to enter data elements not currently in machine-readable format from the Whitespar experimental watersheds and load these data into the database for Web access. The final objective is to develop a Master Watershed Stewards Program to train volunteers to take an active role in educating residents about critical watershed issues.

The original ASCII data files resulting from research conducted on the BCEW are in a format and structure that is difficult for potential users to work with. In fulfillment of the first objective, the files have been imported into Microsoft Access tables, checked for anomalies, and corrected where necessary. There are 11 data categories: sediment, streamflow, nutrients, precipitation, temperature, humidity, wind, snow, multiple resource inventories, range, and timber. The annual and daily streamflow data have been converted and have been used to develop two prototype Web interfaces. The remaining data categories have been prioritized and the files are in the process of being imported into the relational database. The data sets will become available for use as soon as they are in the database.

Two interfaces have been developed to access the data sets. One allows the user to select a data category, time period, gauging station, and output format (Figure 2). The selected data can then be viewed in a simple table or retrieved in a structured ASCII format that is easy to import into a spreadsheet or modeling program for analysis. A preliminary graphing capability of a single variable is provided, and this will be expanded.

The other interface is map based, allowing users to choose the watershed or stream gauge of interest and to view the associated data (Figure 3). This Web-based GIS interface has been implemented using the Environmental Systems Research Institute's (ESRI) ArcIMS Internet Map Server. Map layers for a variety of land features help users locate the area they are interested in and provide visual reference to such things as the effects of the various watershed treatments. The layers currently provided include towns, stream gauges, highways and roads, streams, treatments, watershed boundaries, the BCEW boundary, a 1993 Landsat satellite image, elevation, and shaded relief. This map-based interface is linked to the data interface so the user can select and view data from the same Web site.

In fulfillment of objective 3, an inventory of the information, materials, and data collected on the other Central Arizona Highlands watersheds has been completed and is being used to provide direction for data entry, in objective 4, of the data elements for the Whitespar chaparral watersheds. A draft of the Chaparral experimental watersheds Web site has been prepared with information similar to that developed for the USDA Forest Service Rocky Mountain Research Station's Beaver Creek Evaluation Project Web site (http://www.rmrs.nau.edu/wsmgt/beavercr/). This material and Web site will soon be merged with the project's Managing Semi-Arid Watersheds Web site.

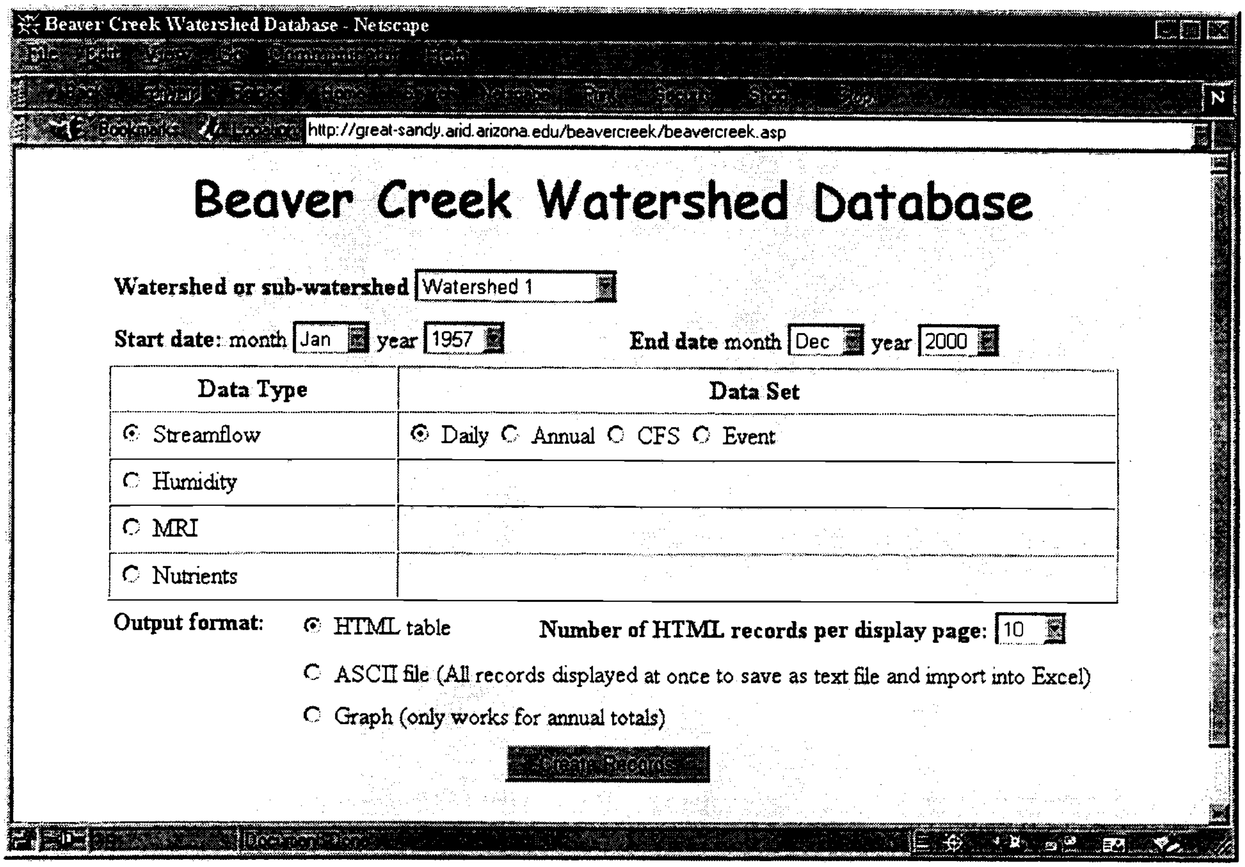

Figure 2. Attribute search interface.

The Master Watershed Steward (MWS) course (objective 5) is under development. This program is currently being marketed to local natural resource management agencies and interested partners to generate support. In addition, a Master Watershed Steward Advisory Committee has been formed to help steer the curriculum development and course planning. Successful Master Watershed Steward programs have been offered in Washington, Ohio, and Alaska. In Arizona, we plan to use the Arizona Natural Resource Wonders Curriculum (Howery et al. 1999) as the core to teach the 8-week course. Supplemental materials will be added to this curriculum as needed. The course will also be augmented by two all-day field trips to see and discuss watershed issues on various sites and have hands-on experience with tools commonly used by watershed managers.

The Master Watershed Steward program is modeled after the Master Gardener Program. After the training is provided, the participants become Associate Master Watershed Stewards. After completing 50 hours of volunteer service, they become Certified Master Watershed Stewards. After finishing the course, Master Watershed Stewards will assist local schools, agencies, and NGOs with projects and educational efforts. Instructional materials developed for the Master Watershed Steward program will be adapted for the Web site. Local partners, such as the USDA Forest Service, Prescott Creeks Preservation Association, ranchers, and schools have indicated that these volunteers will have a positive impact on the local community. Several areas have been identified where these trained volunteers could have a significant impact: riparian stewardship, revegetation efforts, rangeland monitoring, well water quality, noxious weeds, wildland-urban interface issues, septic systems, and youth education. The Master Watershed Steward Program will bring together people of diverse backgrounds and values to get involved in local watershed issues that affect the citizens of Arizona.

Both the Phase 1 and Phase 2 projects have provided a unique opportunity to combine the strengths of three units: the USDA Forest Service,

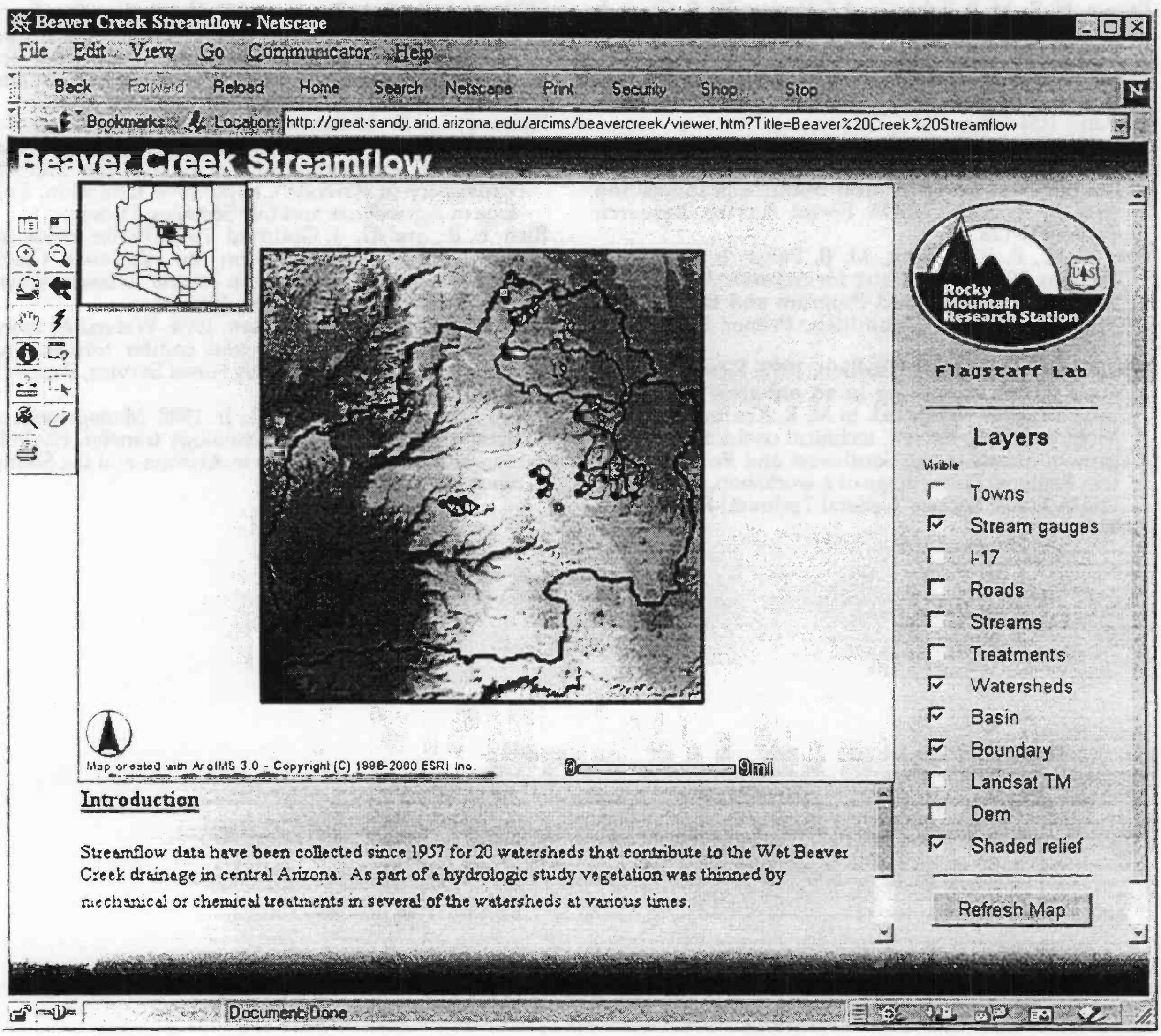

Figure 3. Map-based search interface.

Rocky Mountain Research Station, Flagstaff, Arizona, as a major repository of watershed management information; the University of Arizona Cooperative Extension, with its commitment to training and information dissemination; and the University of Arizona Arid Lands Information Center, for the necessary Web site management expertise.

By bringing practical and field-tested data on watershed management to the World Wide Web, this project provides a value-added service to researchers, practitioners, and educators. In addition, this project seeks to make science more useful, helping the general public make better and more informed decisions concerning the use of their own arid and semi-arid natural resources. In summary, this delivery method facilitates better access to management information and technology, ultimately contributing to the increased sustainability of arid and semi-arid watersheds around the globe.

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THE COST EFFECTIVENESS OF MULTI-OBJECTIVE FOREST MANAGEMENT IN THE WILDLAND URBAN INTERFACE

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Abstract

Many cities and towns in the upland areas of the Southwest are built in the middle of dense forests and woodlands. These forests are prone to catastrophic wildfires, which are major threats to the cities and towns as well as the forest ecosystems themselves. One of these cities is Flagstaff, Arizona. The city is located at 7,000 ft above sea level and at the foot of a 12,500 ft mountain surrounded by the largest continuous ponderosa pine forest in the United States. Because of its location, the city enjoys a four-season climate, with one of the most beautiful landscapes and serene environments. Yet, it faces major threats from catastrophic wildfire and flash flood hazards. The latter may occur in the form of rapid flows down the side of the mountain following heavy rainfall or fast-melting snow events. This paper evaluates the cost effectiveness of minimizing both wildfire and flood hazards while increasing the area's amenity and commodity resources and maintaining its ecosystem integrity in a multi-objective framework. Amenity resources include aesthetics, ecosystem diversity, wilderness, environmental quality, and the area's historical and cultural values, whereas commodity resources are timber, water, forage, and other resources that have economic value. These different management objectives, and the various groups with stakes in the condition of the wildland-urban interface, are considered in determining the most cost-effective forest resources management scheme.

Introduction

Continuous growth and encroachment of cities and other urban communities into their surrounding wildland ecosystems is creating a new environment known as the wildland-urban interface (WUI). The wildland-urban overlap usually develops into a unique ecosystem with its own ecological, socio-economic, and safety issues. The uniqueness of these issues is based on climatic and biophysical characteristics as well as the affected communities' societal norms and economics. A significant number of the 65,000 residents of Flagstaff live in the WUI, which is characterized by a four-season climate consisting of warm summers, cold winters, and moderate periods between. Precipitation is almost equally divided between summer and winter. The vegetation consists mostly of ponderosa pine forests, which have become dense from wildfire suppression over the last century (Covington et al. 1997). Residential home developments and recreational facilities are encroaching heavily on some parts of this forest.

The interface of the ponderosa pine forest ecosystem with the expanding urban development, frequently recurring drought periods, and recent near-miss wildfires in the WUI have become cause for major concern in the community. This concern has led to a partnership of various interest groups whose aim is to develop sound ecosystem management in the WUI, to protect the community and the forest from catastrophic wildfires, and to promote diverse and healthy ecosystem conditions (Hill 1998; U.S. Forest Service and Grand Canyon Forests Foundation 1998). However, a number of other objectives also influence the management of the WUI, including developing and maintaining recreation opportunities, increasing or protecting aesthetic quality, preserving historical and cultural resources, optimizing operational and maintenance costs, and reducing the chances of flooding. This paper addresses these issues in a framework of cost-effectiveness analysis.

Problem Statement

The wildland ecosystems in northern Arizona have been exploited for their market-oriented commodities over the past century, mainly livestock grazing and timber harvesting, primarily old growth. These kinds of uses in combination with

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the exclusion of the natural fire regime have led to serious degradation of these ecosystems (Covington et al. 1997). Frequent low-intensity wildfires had been natural in the ponderosa pine forests and other ecosystems in northern Arizona. Overstory and understory vegetation, as well as the other components of the ecosystem, have adapted to this regime. Low precipitation rates cause dead woody debris to decompose very slowly. Under natural conditions, there had never been a large accumulation of fuel in this forest system. Further, the natural fire regime kept the regeneration of new trees in balance with the dying of old trees. However, government policies on fire suppression disrupted this normal process and brought tree density to dangerously high levels. The situation created the right circumstances, in the form of fuel accumulation on the forest floor and ladder fuels, for catastrophic wildfires to occur easily and more frequently than they would under normal conditions.

According to Hill (1998), the USDA Forest Service spends billions of dollars each year to suppress wildfires. Moreover, the amount spent on fire suppression each year has steadily increased with the increase in the size of forests to burn (Hill 1998). Hill's work was published before the severe fire season of 2000, which had been the worst, in terms of loss and costs, in more than 50 years (Associated Press 2000). Hill further argued that it would be less expensive to treat every acre of the 70 million acres of pine forests in the interior West with a combination of mechanical removal of vegetation and accumulated fuels and prescribed fire than to continue to pay for the increasing cost of suppressing wildfires. Preventive treatment of the 39 million high-risk acres at an average cost of \$320 per acre would still be substantially less expensive than to continue suppressing wildfire in the West. His calculation does not take into consideration any non-market value lost in the fires or gained by restoration treatments. Under these circumstances, the monetary figures that Hill presented to Congress in his GAO report should be considered modest. Nonetheless, the estimated average cost per acre for prescribed burning in the Southwest is only \$40.22 in 1994 dollars (Cleaves et al. 2000).

Fuel reduction efforts, as described by Hill, have been conducted in the Flagstaff WUI on a very small scale. The efforts have remained relatively small in part due to the legal intervention by environmental groups concerned with renewed exploitation of natural resources under the umbrella of forest health (Center for Biological Diversity 1999). However, restoring and sustaining ecosystem health have enhanced various nonmarket values, such as wildland ecosystems, especially in the WUI (Kim 1999). Prescribed burning, in particular, was found to enhance the recreational experience (Shortess 1986).

Cost-Benefit Analysis Versus Cost-Effectiveness Analysis

Cost-benefit analysis has traditionally been used to quantify and rank different alternative projects, management activities, or any social or economic benefits on the basis of a set of criteria to help decision makers choose the most preferred alternative course of action (Weaver et al. 1982). However, economic appraisal of environmental policies and projects using this method is problematic, because some benefits cannot necessarily be measured in monetary terms (Macmillan et al. 1998). It is difficult to put market values on nonmarket goods and services. For example, it may be possible to figure out costs of maintaining and enhancing aesthetic quality and the preservation of historical resources (in terms of equipment, fuel, labor, and other costs), but it would be difficult to determine the benefits of these costs, if we don't know the real value of the resources we are attempting to protect and preserve. There are many indirect costs and intangible benefits involved, which are not estimated by normal market values, ranging from soil compaction to loss of biodiversity (Pinjuv et al. 2000). A more useful approach to evaluate such problems is using cost-effectiveness analysis.

Cost effectiveness is defined as a technique for choosing the most preferred course of action among given alternatives in terms of its cost and effectiveness to attain specified objectives (Livingstone and Gunn 1974). The evaluation process in this technique includes articulating project management objectives, listing all alternative actions, and specifying criteria to measure the effectiveness of the alternative actions to achieve the desired objectives. The advantage of this method is that it does not rely on monetary valuation, but rather selects projects on the basis of various criteria including nonmarket cost effectiveness in achieving the predetermined objectives (Macmillan et al. 1998). In land resources management practices, costs are usually considered together with environmental impacts and the use of available technology (Bureau of Land Management 1991). Costs are measured in the same manner as in the cost-benefit analysis method, in terms of use of equipment, manpower, and other resources, but benefits are assessed in terms of their effectiveness, where effectiveness is a comparative measure of the performance levels of different alternatives to achieve a given set of objectives (Pinjuv et al. 2000). When using alternatives that incur similar costs to achieve ecological objectives, cost effectiveness can be applied to select the alternative with the least negative environmental, social, cultural, and other impacts as the most effective solution. Yet another advantage of this analysis lies in its systematic and transparent use of judgment (Quade 1967). It allows decision makers to follow any analytical assumptions whether they agree or disagree and draw their own conclusions. The limitations of the cost-effectiveness analysis lie in selecting the measure of effectiveness. Because some of the attributes may not be directly measurable, the method is vulnerable to imperfect information and judgment (Livingstone and Gunn 1974).

Cost-Effectiveness in the WUI

Cost of Treatments

The application of cost-effectiveness analysis to land resources management (dollars per acre) is normally achieved by consolidating several individual projects into one (BLM 1991). For example, a recent study by Pinjuv et al. (2000) that evaluated the ponderosa pine ecosystem restoration in the Flagstaff WUI found cost-effectiveness analysis to be a "useful tool for optimum harvest recommendations." Further, it was calculated that depending on stand density, the treatment cost of merchantable and pre-commercial thinning ranged from \$125 to \$460 per acre. This cost estimate took into consideration the hand cutting of trees with a diameter at breast height of 5 inches and smaller. The hand cutting method has fewer negative environmental effects, such as soil compaction and damage to remaining trees, than machine harvesting, making it a more cost-effective method. The cost-effectiveness value for different harvesting methods in the Flagstaff WUI is evaluated by dividing the harvesting cost by the percentage of undamaged trees left in the ground. A higher cost-effectiveness value represents a more efficient method (Table 1), provided that the harvest cost also accounts for environmental and other non-market values, or the percent undamaged includes environmental ecosystem health, cultural, amenity, and commodity values.

Table 1. Example of cost-effectiveness of different treatment methods in the Flagstaff WUI. Values exclude old growth harvesting (taken from Pinjuv et al. 2000).

Harvesting Method	Harvest Cost (\$/Ha)	Cost (% undamaged) (\$/%)	Cost Effectiveness
Whole tree	1098	1098/65	16.9
Hand harvest	1497	1497/95	15.8
Cut-to-length	1297	1297/80	16.2

The cost of prescribed fires, however, is less than that of harvesting. Based on a survey of 10 years of prescribed burning on Forest Service lands, Cleaves et al. (2000) found that the per acre costs of treatment in the Southwest vary from one type to another. For example the cost for slash reduction is \$77.05 per acre. Prescribed burning costs \$38.85 per acre for management ignited, \$7.67 per acre for natural, \$37.30 per acre for brush, range, and grasslands, and \$40.22 per acre for all types together. But the survey also mentioned that the risk to structures and property damage was higher in the Southwest than in other parts of the country.

Effectiveness for Recreation

The steadily increasing number and high diversity of recreational users of forest lands have led to some social and managerial conflicts, both between recreationists and other forest uses such as mining, grazing, and timber harvesting interests, and among different recreationists themselves (e.g., motorists vs. non-motorists). According to the Multiple-Use Sustained Yield (MUSY) Act of 1960 and the National Forest Management Act of 1986, national forest lands are expected to be managed for multiple uses, including livestock grazing, maintaining wildlife habitat, water yield, timber production, recreation, and even scientific and educational purposes (Richards and Daniel 1991). However, with a rapidly growing population in Arizona as well as southern Nevada and southern California, the recreational activities in the forested mountains of Arizona have become more important than some of the other more traditional forest uses (Bureau of Business and Economic Research 1993). Timber harvesting, once an important source of income in Flagstaff, now employs less than 30 people in the county (High Desert Industries, personal communication 1999). Livestock grazing activities have been reduced dramatically in recent years (M. Lee, Northern Arizona University, personal communication

2000) and mining operations in the area are also coming to an end due to public pressure.

Today, mostly due to northern Arizona's unique physiographic features, a four-season climate, and scenic forest ecosystem, tourism is the major source of employment in Flagstaff (Arizona Department of Commerce 1998). Hence, multi-objective management of the forest system in the WUI should consider enhancement of recreational opportunities as one of the important management objectives. The USDA Forest Service has developed a recreational opportunity spectrum to systematically assess the recreational capability and potential of a management unit through inventory of recreation opportunity settings and a managerial framework (Richards and Daniel 1991).

Regardless of the recreational setting, the entire forest spectrum would benefit from restoration and fire prevention treatments. A catastrophic forest fire would render the forest aesthetically very unattractive for recreational activities. However, restoring the forest would help to prevent the occurrence of a major forest fire and would enhance recreation and other commodity and amenity resources in the WUI. Under existing conditions, the rising number of individuals recreating in wildland environments increases the chances for a catastrophic wildfire to occur, for example the "Leroux" fire in June of 2001 just outside of Flagstaff.

Effectiveness for Aesthetic Value

Seven million visitors pass through Sedona and Oak Creek Canyon, located just south of Flagstaff, each year (Stafford 1993) and many of these visitors come to enjoy the area's scenic beauty. This was demonstrated by Daniel and Boster (1976) who developed a method to estimate people's perceptions of scenic beauty. They called it the scenic beauty estimate (SBE). The SBE for a ponderosa pine forest is based on aesthetic values of different levels of timber stands expressed in square feet of basal area per acre. Applications of the SBE method in northern and central Arizona found that the general public prefers a forest density of 80–140 sq ft per acre of basal area (Tecle et al. 1998; Figure 1). The current level of average stand density in the forest around Flagstaff is about 200 sq ft per acre basal area (Tecle et. al 1998). However, restoration and fire prevention treatments can reduce the density of the forest to approximately 85–120 sq ft per acre of basal area depending on individual treatments. To be most effective, these treatments should include mechanical removal and prescribed burning (Hill 1998). Taylor and Daniel (1984) found that in addition toreducing fuels, light burning may also improve the scenic beauty of forest landscapes over time, whereas severe burning has a negative impact on scenic beauty.

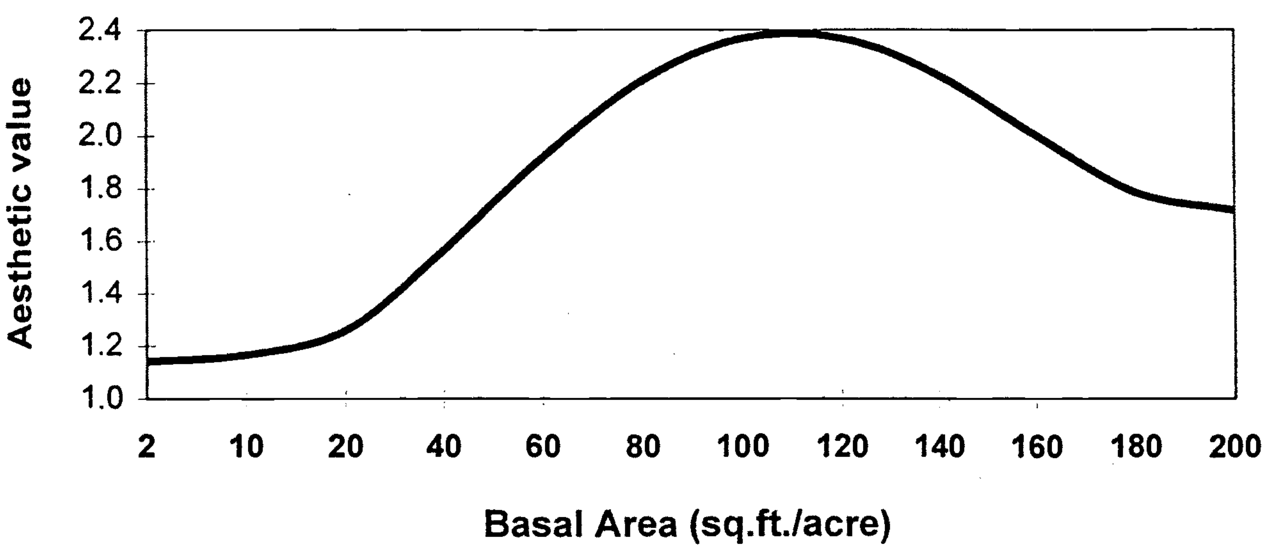

Figure 1. Forest aesthetic value response to changes in residential tree basal area (Tecle et al. 1988).

Effectiveness for Historical Resources

Despite the fact that Flagstaff is only 120 years old, it has many historical resources. The dry climate preserves artifacts and ecofacts quite well (Price and Feinman 1997). Ecofacts are any remains of plants, animals, sediments, or other unmodified material that results from human activity. Flagstaff and its historic sites would be at risk if a catastrophic wildfire were to erupt southwest of the town. Simulation results using fire area simulator-model development and evaluation (FARSITE; Finney 1998) showed that a fire starting in the forest southwest of town would spread rapidly into the city with the help of prevailing winds.

There are also many Native American historical and cultural resources in the greater Flagstaff area (USDI National Park Service 1995). The region was home to the Anasazi, Hohokam, and Sinagua Indians from approximately 850 to 1400 A.D. (Price and Feinman 1997). These tribes have left their traces, which are important to preserve. In this case, fire may not be the major threat to the preservation of these historic and cultural sites, but people are (Fink 1998; L. Farnsworth, USDA Forest Service, personal communication 2000). Curious recreationists, hobby archeologists, and pottery collectors have done the most damage to Indian cultural sites (USDI National Park Service 1999). In this instance, the most efficient management decision is to keep such people away from sensitive sites. This has been done successfully on the San Francisco Peaks and several other areas in and around Flagstaff (M. Lee, personal communication 2000).

Effectiveness for Optimizing Operational and Maintenance Costs

One source of problems with preserving Indian cultural sites is roads. There have been several instances when road construction teams have mistakenly excavated Indian burial sites (L. Farnsworth, USDA Forest Service, personal communication 2000). Thus, new roads being constructed or existing roads being expanded on U.S. Forest Service lands now require archaeological evaluations to determine that projects do not have significant effects on cultural sites (L. Farnsworth, personal communication 2000). Similar regulations apply to timber harvesting sites and other developments in accordance with the National Environmental Policy Act of 1969 (Vogt et al. 1997). Another way of optimizing operational resources is to reduce the number of roads in the forest around Flagstaff. The overall dry weather followed by intensive thunderstorms causes roads to degrade very quickly, and they require a great deal of maintenance.

Another important aspect of optimizing operational and maintenance costs is to ensure that forest restoration or fire prevention treatments in the WUI are designed not only to protect the residential homes within the interface as well as the city proper from a catastrophic wildfire, but also to protect the wildlands from fires that start in the residential areas.

Effectiveness for Flooding

A large portion of the city of Flagstaff, including about 70 percent of its Historic District, is built in the floodplain of the Rio de Flag (U.S. Army Corps of Engineers 1999). In spite of this, the risk of flooding is a factor that is often overlooked in Flagstaff's WUI activities.

The original settlement of Flagstaff was located a couple of miles east of the current downtown. But after that settlement burned down twice, in 1882 and 1884, the town was rebuilt around the new train depot, which was located on the Rio de Flag floodplain close to the stream because the latter provided easy access to water for the steam locomotives passing through the city. As the town continued to grow, development occurred generally by expanding outward from the city center, within the topographic depression around the train depot. This area was and still is subject to inundation from Rio de Flag flooding events. The continued development of the land resulted in rerouting of the natural Rio de Flag channel and further encroachment into the floodplain (U.S. Army Corps of Engineers 1999). Even though minor floods were recorded in Flagstaff's early history, the transient population in those early years was oblivious to any potential flooding and the area continued to develop and became more densely populated. There have been a total of 17 recorded floods in Flagstaff since 1888 (U.S. Army Corps of Engineers 1999). However, development in the floodplain continued until the Federal Emergency Management Agency's (FEMA) flood insurance policy was adopted in 1983. Since then, any structural development within the floodplain has been required to have its base elevated above the 100-year floodplain zone. More than 100 years of unregulated development in the floodplain has left the Rio de Flag channel very narrow and shallow throughout its course in the city. In contrast, upstream and downstream of

Flagstaff the natural channel of the Rio de Flag is very wide and deep and is surrounded by a dense forest. Though residential and commercial development varies from light to heavy along the tributaries, it is extensive along the banks of the main Rio de Flag channel passing through town.

There is a major risk in the extensive development of the Rio de Flag floodplain in Flagstaff. Therefore, in the event of a major flood occurring, there would be substantial structural damage and economic loss throughout this portion of the city. Possible structural types that would sustain damage include historic properties, public infrastructure and services, parts of Northern Arizona University, and the Burlington Northern & Santa Fe Railroad east-west main line. At present, nearly half of the 100-year floodplain along the Rio de Flag is zoned as residential and almost a quarter as commercial. In the event of a major flood, transportation problems would make a large portion of the city inaccessible for a few days.

Further, if Flagstaff does not protect itself and prepare for a major flood, it will continue to be at risk and may suffer substantial economic, social, and environmental damage in such an event. Approximately 1,500 existing structures, worth $385,000,000, could suffer about $93,000,000 worth of damage from a 100-year flood event without considering possible damages to historical structures (U.S. Army Corps of Engineers 1999).

The possibility for a peakflow flood event would increase significantly after restoration or fire prevention treatment in the WUI, as demonstrated in the Beaver Creek experimental watershed south of Flagstaff (Brown et al. 1974). However, the effects of treatments on flood peak may not last long as herbaceous and tree seedlings start to cover the treated landscape, slowing surface runoff and dissipating peak flows within a short time. Even though there may be substantial costs in property damage from a severe flood event within the city of Flagstaff, the damage caused by a catastrophic wildfire in the WUI may potentially be higher, depending on the location and size of the fire. The damage from forest fires may include losses to wildlife habitat, recreational areas, aesthetics, and property damages, especially in areas located outside the city limits.

Conclusion

The prevention of catastrophic forest fires is one of the major objectives of a multi-objective forest management plan in the wildland-urban interface around Flagstaff, because almost all other objectives are in one way or another affected by the threat of a wildfire or any restoration or fire prevention treatments. Management issues influenced by such treatments are recreation opportunities, aesthetic quality, historical and cultural resources, operational resources, and flooding. Recreation and aesthetics are two management objectives that would suffer most severely from a burned forest ecosystem, which would have adverse effects on the tourism economy in northern Arizona. However, the effects of prescribed burning to reduce catastrophic wildfires may have insignificant impacts on recreation and the area's scenic beauty. Operational resources would also be saved, because the preventive measures are by far less expensive and more effective than reactive measures, such as fighting wildfire and rehabilitating burned areas. Although a reduced stand density might make the WUI around Flagstaff and the city itself prone to flooding, the cost of a catastrophic wildfire is potentially much higher than damages from flooding. In short, restoration, fire prevention treatment, or both in the WUI seems most cost effective, especially if the treatment level is carefully selected to meet the other forest ecosystem management objectives, including minimizing the flood hazard.

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UPDATING HYDROLOGIC TIME-TREND RESPONSE FUNCTIONS OF FIRE IMPACTS

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Fire has historically been part of the natural environment of the southwestern United States. It is not surprising, therefore, that the frequent occurrence of wildfire and prescribed burning in the region has led to a useful body of knowledge and experience concerning the effects of fire on the resources of natural ecosystems. Nevertheless, the region's recent fire history has reinforced the need to learn still more about the effects of fire on ecosystem resources. Postfire changes in plant, animal, and soil and water resources take place at the time of, or shortly after, a fire occurrence (Whelan 1995; Bond and van Wilgen 1996; Pyne et al. 1996; DeBano et al. 1996, 1998). The magnitudes and durations of these changes can often be estimated when quantitative data are available. However, sufficient knowledge necessary to obtain reliable estimates for other than site-specific conditions is seldom available and, when this is the case, estimates of postfire changes in ecosystem resources must be made in the context of the data and information that are available or can be made available at reasonable cost. This paper outlines background issues and a plan to estimate postfire changes in ecosystem resources by updating the time-trend response functions of fire impacts.

Time-Trend Response Functions

Estimating postfire changes in ecosystem resources using time-trend response functions involves interpretations of the flows of damages to the resources and/or benefits for the resources through time since a fire occurrence, within a framework of both a physical and economic analysis of the postfire conditions (Lowe et al. 1978; Ffolliott et al. 1987; Ffolliott et al. 1988; DeBano et al. 1998). Some combination of on-site measurements and the experience and judgment of managers, fire behavior specialists, and ecologists often suffices for approximating the form of a time-trend response function for a resource and fire severity.

Time-trend response functions represent postfire changes of a resource in relation to the time since (a) a fire of a known severity or (b) a range of fire severities; that is, a set of time-trend response functions. A study being planned with scientists of the Rocky Mountain Research Station, USDA Forest Service, will focus on updating both forms of time-trend response functions with data and other information made available since their original formulations. The general framework for either developing or updating time-trend response functions is as follows:

- Resource values for postfire conditions are obtained by either (a) sampling the attribute at different points in time after a fire has occurred or (b) sampling the attribute on a series of burned areas representing fires of similar severity but varying fire histories. (Burned areas forming the data and informational base for the original formulation of the hydrologic time-trend response functions are included in the sampling.)
- The postfire values are then compared to corresponding resource values obtained by sampling unburned (control) areas. Assuming that the resource in question responds in the same manner to fluctuations in weather conditions, time of year, and cyclic alterations, differences in the two sets of values are considered to be indicators of changes in the ecosystem resources due to fire only.
- The differences in the two sets of values are shown as either (a) a stream of ratios over time between the resource values obtained for the burned and unburned areas or (b) absolute changes (either increases or decreases) in resource values over time following a fire in relation to the unburned area. Applying the ratio from function (a) to unburned values enables a manager to define a time-stream of postfire ecosystem resource values. This

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application provides a basis for calculating the sum of discounted fire-caused responses that are expected to occur over time (see below). The changes reflected in response function (b) permit the same kind of calculation for the resource to which it is applicable (DeBano et al. 1998). Time-trend response functions developed for different fire severities incorporate additional information, further sharpening the manager's ability to estimate fire impact over time.

Streams of annual ratios representing time-trend response functions can be converted to fixed-term annuities that represent equal annual returns from the resource of concern. Annuities are most commonly considered in monetary terms; however, the concept is equally applicable to non-monetary flows (Lowe et al. 1978; Ffolliott et al. 1988). Annuities allow the annual stream of ratios to be condensed into a single annual index value. Theoretically, an annuity value of 1.0 is "indifferent" to the stream of annual ratios (no change in postfire responses), an annuity value greater than 1.0 represents increases (positive postfire responses), and an annuity value lower than 1.0 indicates losses (negative postfire responses).

An Example: Estimating Postfire Changes in Streamflow

Figure 1 shows a time-trend response function illustrating a stream of ratios for changes in annual streamflow volumes in relation to time since a fire of a known severity in southwestern ponderosa pine forests. This function was developed by plotting ratios of the resource values obtained by sampling on burned and unburned areas (Ffolliott et al. 1987, 1988). Interpretation of this time-trend response function is illustrated through an example in which the effect of a fire is examined in terms of its effect on annual streamflow amounts. A moderate fire intensity (17,500 to 35,000 kilojoules/meter/second) with an average flame length of one-third to two-thirds of a meter is selected to characterize this hypothetical fire. It is also assumed (perhaps unrealistically) that the fire burned uniformly over the forest. The impact of the hypothetical fire on annual streamflow is examined for a postfire evaluation period of 10 years and (arbitrarily) at a 5 percent discount rate.

(A discount rate determines how much weight is given to the different annual ratios representing time-trend response functions. The greater the discount value, the more heavily future ratios are discounted. For example, if a 5 percent discount rate is used, ratios for 1 year after a fire are weighed 2.5 times as heavily as ratios for 20 years following a fire. If a 10 percent discount rate is used, however, ratios for 1 year after a fire are weighed more than six times as heavily as ratios for 20 years after the fire.)

Hydrologic changes that take place on a site after a fire has occurred can contribute to changes in annual streamflow amounts (Krammes 1990; Ffolliott et al. 1996; and others). Variables that affect both cumulative infiltration in time and infiltration capacity can be affected by fire to varying degrees, often adversely, resulting in decreased infiltration, increased overland flow, and (ultimately) increased streamflow amounts (Pyne et al. 1996; Brooks et al. 1997; DeBano et al. 1998). Additionally, considering rainfall events, the reduced forest density and litter cover following a fire and the possible occurrence of hydrophobic soils can decrease evapotranspiration losses, causing larger streamflow amounts. During winter months, however, a reduction in forest overstory caused by a fire can allow a greater proportion of a snowpack to be lost to evaporation processes, a phenomenon that results in less streamflow amounts. Nevertheless, annual streamflow amounts (considering both rainfall and snowmelt events) in southwestern ponderosa pine forests are generally increased by burning, at least in the first 20 years after the fire. The change in annual streamflow amounts (in terms of an annuity value) is 3.2 for the initial 10 years after the hypothetical fire; that is, annual streamflow amounts for the evaluation period will be 3.2 times that of the prefire annual streamflow amounts. This increase is attributed to the large increase in streamflow in the years immediately after burning.

Updating Hydrologic Time-Trend Response Functions

The plan for this collaborative study is to update time-trend response functions that were originally developed for annual streamflow volumes, suspended sediment concentrations, and nutrient, heavy metal, and other resources in southwestern ecosystems (Ffolliott et al. 1988). Burned areas with known fire histories and adjacent unburned areas, including areas forming the basis for development of the original time-trend response functions, will be evaluated or re-evaluated in terms of the attributes selected for updating and

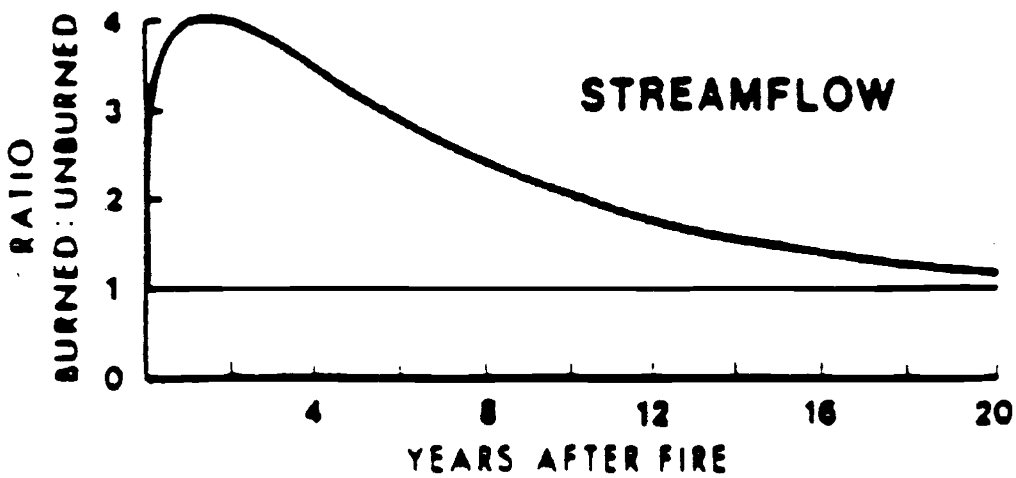

Figure 1. A time-trend response function shown as a stream of ratios for changes in annual streamflow amounts in relation to the occurrence of fire of a known severity in southwestern ponderosa pine forests.

expanding. The field measurements, observations, and monitoring necessary to determine postfire changes in the attributes and the analytical procedures required to formulate the updated time-trend response functions will be largely similar to those specified for earlier studies (Lowe et al. 1978; Ffolliott et al. 1987). Literature reviews of fire effects on hydrologic and other ecosystem resources will also help to structure time-trend response functions for the resources to be studied. The personal experiences and professional judgments of managers, fire behavior specialists, and hydrologists will be solicited in refining the forms of the time-trend response functions.

Initial emphasis will be placed on obtaining a better understanding of the factors that largely dictate the form or that could cause changes in the form of a time-trend response function. These factors include fire characteristics (such as intensity and severity), vegetation type, and soil characteristics. These and other factors considered individually, but more often in combination, affect the magnitude, rate of rise and recession, and length of time to attain levels representing prefire conditions for a hydrologic resource. The paired burned and unburned areas (watersheds) to be evaluated will be selected and evaluated in terms of these factors.

Revision of Computer Program

The updated time-trend response functions will be incorporated into the computer program BURN (Ffolliott et al. 1988) to expand the simulator's applications in terms of the ecosystem resources simulated, vegetative types considered, and fire intensities and severities confronted. The computer program will retain its original modular format for ease of future updating of the simulator, with a user-friendly interface for data entry. The revision of BURN, a user's manual, and examples of its application will be made available to interested people through publications, CD-ROMs, and the Internet.

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ASSESSMENT AND RECOMMENDATIONS FOR STREAM RESTORATION PLANS IN CLOVER SPRINGS VALLEY, ARIZONA

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Many stream morphologies exist along the Mogollon Rim of northern Arizona. Of these, wet meadows are unique and critical for the health of the area's stream ecosystems and ponderosa forests. Many of these meadow systems have been incised due to recent anthropogenic and natural activities. In some instances the cause of channelization remains unclear. It is possible that regional incision may have occurred primarily in response to climate change or extreme runoff events tied to high seasonal precipitation. Channel incision lowers the local water table, threatening the wet meadow riparian ecosystem. In addition, increased erosion reduces downstream water quality. One such degraded system in the Clover Spring Valley has already been incised up to 3 m. Anthropogenic activities, such as straightening of the stream channel, have severely impacted the morphology, hydrology, and biology of the stream system in this area. In an effort to restore the health of this system, an interdisciplinary group from Northern Arizona University, in conjunction with the National Forest Service, has developed a demonstrative plan that includes restoration of the stream's natural meanders and removal of a Forest Service road. Previous efforts to alleviate channelization in Clover Spring Valley that involved the construction of check dams have failed. Determination of the "correct" pre-disturbance morphology and the long-term stability of the restoration project has yet to be addressed. This review examines the efforts taken to determine the correct pre-disturbance morphology and evaluates the proposed techniques and strategies for restoration.

Introduction

Part of the headwaters to the Verde, Little Colorado, and Gila Rivers are upland wet meadows along the Mogollon Rim of central Arizona approximately 80 km south of Flagstaff. Many of these wet meadows have been degraded as a result of stream channel incision, sometimes to bedrock. The Verde River Headwaters Riparian Restoration Demonstration Project is designed to restore the wet meadow ecosystem along a reach of stream in the Clover Springs Valley. We outline and evaluate the strategy and methods proposed in the restoration project and make recommendations for a more sustainable, long-term restoration plan.

Regional Setting

The Mogollon Rim is part of an escarpment that bisects Arizona diagonally from the northwest corner to the central-eastern part of the state. This escarpment is approximately 500 km in length and forms the boundary between the Colorado Plateau to the north and the Transition Zone–Basin and Range provinces to the south. The escarpment is bounded on its western margin by the structurally controlled Grand Wash Cliffs and on the eastern margin by the White Mountains. The Mogollon Rim has local relief of up to 600 m (Peirce et al. 1979). The rim forms a drainage divide that separates the northward-flowing streams that feed the Little Colorado River from streams to the south that drain into the Verde and Salt Rivers.

Background

Quaternary deposits along the Mogollon Rim contain abundant proxies for interpreting the region's paleoenvironmental record, including sediment and midden deposits. Pollen, macrobotanical, and fossil remains derived from stratigraphic deposits in lakes, bogs, alluvium, and caves, as well as deposits of packrat middens and herbivore dung provide evidence for late Quaternary environments. Regional climatic oscillations can be deduced from these assemblages (Anderson et al. 2000). As a result, numerous studies have established an extensive 50,000 yr climate and

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paleoecological record for the Mogollon Rim region (Anderson 1993; Anderson et al. 2000; Hasbargen 1994; Whiteside 1965).

The southern Colorado Plateau contains the highest concentration of studied paleoenvironmental sites, and also contains the most complete paleoenvironmental records (Anderson et al. 2000). In compiling previous paleoenvironmental studies, Anderson et al. (2000) cited evidence for periods of wetter and colder climates during the middle and late Wisconsin. Anderson (1993) followed up a study by Whiteside (1965) on the paleoecological record beneath Potato Lake, which is located on the Colorado Plateau in central Arizona. Anderson's (1993) findings concurred with the interpretation of a wet and cool mid and late Wisconsin climate. Hasbargen's (1994) research on nearby Stoneman Lake sediments suggested a pattern of strong monsoons between 10,000 and 8,000 yr B.P. that subsequently tapered off to warmer, drier conditions. The Arizona monsoon season's sensitivity to glacial intervals (Anderson et al. 2000), coupled with the orographic rise of the Mogollon Rim, could have induced major geomorphic responses associated with these climate changes (Diana Anderson, personal communication). As a result, the deviations in climate should have triggered distinct fluvial geomorphic responses throughout the region.

Kennedy (1999) completed a geomorphic analysis of a stream system located at Clover Spring that underwent stress induced by anthropogenic and climatic effects. The studied reach is representative of streams and wet meadows found across the Mogollon Rim. The focus of the study was to determine the causes and timing of aggradation and erosion that occurred along a section of stream within the Cover Springs stream system. One goal was to determine the cause of a regional incision event that occurred during the 1880s, which was previously attributed to climate, overgrazing, and channel straightening. The depositional record of the stream extended 7,000 yr B.P., but the study focused primarily on the late nineteenth century incision event. Radiocarbon dating, dendrochronology, and historic photos were employed to determine the timing of preserved depositional events. Tree rings were examined to distinguish wet and dry periods, however the range of the tree rings was limited to slightly over 100 yr B.P (Kennedy 1999). Kennedy was able to correlate a wet period evident in tree rings to the regional incision event. In addition, grain size analysis of the stream deposits suggested that depositional environments were similar through the time period studied. Kennedy (1999) concluded that the coincidence of drought and grazing prior to a wet period during the 1880s contributed to the incision of the stream at Clover Spring.

Insight into the response of fluvial systems to climate change is an important contribution to our understanding of paleoenvironments. Additionally, this information can be valuable to restoration projects that require distinction between natural cyclic changes in fluvial systems and anthropogenic sources of change. For example, heavy grazing of a floodplain adjacent to a stream can produce a considerable decrease in vegetative cover, which can destabilize the stream and facilitate incision. Conversely, a period of incision could be attributed to an interval of high precipitation and consequential runoff following a dry period. As a result, the condition of the stream in question may be well within the natural cycle of the fluvial system.

Project Objectives

The primary goal of the Verde River Headwaters Riparian Restoration Demonstration Project is channel stabilization and re-establishment of riparian wet meadow function along an approximately 8300 m reach of Clover Springs wet meadow. To reach this ultimate goal, three main objectives were formulated: (1) designing and executing a plan to stabilize the channel and preserve the wet meadow in the Clover Springs reach; (2) interpreting the factors leading to incision and deciphering the timing of these events with the intention of creating a long-term mitigation plan; and (3) producing informational material to transmit activities of the project to the public.

To accomplish the goals of objective two, a geomorphic analysis was proposed to determine the mode of sediment deposition and erosion within wet meadow fluvial systems of the Mogollon Rim region. Particularly, the mechanism, timing, and effects of the local incisional events and subsequent depositional events will be evaluated. Supporting objectives are to (1) establish the nature of the system prior to incision; (2) identify various fluvial regimes, identify buried soils, and collect material suitable for ^{14}C dating; and (3) compare paleoclimate records to prehistoric geomorphology. In addition, historic climate records and land-use changes will be compared with the historic geomorphic section to determine how both climate and land-use changes have affected the erosional and depositional history of the area.

Proposed Site Resoration Methods

Active restoration techniques, including road removal, back-filling, and channel diversion, are the proposed methods for restoration. Use of these restoration techniques and determination of the correct channel morphology are based on quantitative measurements and comparisons to similar, non-degraded reference streams. Post-restoration changes will be monitored and compared to pre-restoration conditions.

Pre-Restoration Site Characterization

Pre-restoration site conditions were determined and documented by the production of a high-resolution topographic map in c. 1999. This map was constructed on a 1:3000 scale with the use of a Topcon 310 total station. Coordinates were collected for 2600 survey points and plotted using ArcView. This survey map provides a datum from which post-restoration changes can be measured.

Determination of Correct Channel Morphology

Correct channel morphology has been determined for the project area based on site information and comparisons to stable, non-degraded stream channels found with similar characteristics (reference sections). Site information includes stream profiles and cross sections, hydrologic analysis, and aerial photograph interpretation. This information was combined with observations from the reference sections to attain a stream channel design. Hydraulic analysis was then completed on the newly designed stream channel to further refine the design. Project design is restricted by the base level control imposed by culverts in the upper section of the project area and bedrock located in the lower section.

Restoration Work to Be Completed

Restoration work at the site will occur in two sections. Work on the upper section will include removal of a Forest Service road adjacent to the stream, which will increase the bottom width of the channel from 2.5 m to approximately 5 m. This will lower bank-full velocities and allow the stream channel to become more sinuous. Back-filling of material in the headcut locations will also be done to smooth the stream profile and reduce headward erosion. Other related work on this section will include cutting back the existing stream bank to a more stable profile and re-vegetation of newly exposed sections.

The second, lower section of restoration will include abandonment of the existing incised stream channel and the construction of a new channel. This technique will promote the conversion of an F or G type channel to a more optimal C or E channel configuration, following Rosgen's classification scheme (Rosgen 1996). The new channel will be constructed on the existing floodplain where it should evolve from a sinuous, low entrenchment, gravel-dominated C-5 stream type to a stable, low to moderately sinuous, E-5 stream with gentle to moderately steep channel gradients and low width to depth ratios.

Monitoring Plan

Site conditions will be monitored after the restoration work has been completed to assess the stability of the redesigned channel. Monitoring will include two sets of measurements taken annually for the duration of the project funding and at least once every 3 years thereafter. The two sets of measurements to be taken include a longitudinal stream profile and cross-sectional profiles that will be taken after the stream's annual peak discharge. Other parameters to be monitored include shallow groundwater, vegetation, and surface water flow, as well as repeat photographs of the stream and floodplain. These measurements will allow assessment of stream stability and document any changes in the system, which will allow for a better future understanding of how streams respond to this type of restoration work.

Evaluation of Restoration Strategy and Methods

Proposed as a riparian restoration effort, in contrast to a more extensive watershed restoration, the Verde River Headwaters Restoration Demonstration Project will attempt to restore the function of a wet meadow. An active restoration plan was therefore devised; however, after reviewing the plan, two major suggestions are apparent. First, the initial proposal identified a need for regional geomorphic survey of similar wet meadow–stream systems. Correlation of the geomorphic stratigraphy with existing paleoclimatic research would reveal what responses to climate and environmental change these systems have achieved during the Holocene. With this knowledge, a more comprehensive understanding of the dynamics of these systems along the Mogollon Rim would be gained. Unfortunately, this regional assessment was not completed prior to the conception of the active restoration plan.

A second suggestion is to assess the watershed as a whole. The small stretch of the stream system to be restored during this project is bound by artificial base levels. Culverts create the base level upstream of the project, and upon completion of the restoration project, a weir will create the downstream base level. These controls will preserve the restored section for the short term, but the long-term stability of the reach depends on a broader watershed-based approach to stream restoration.

Many controls contribute to the channel morphologies found within a watershed. These controls include channel gradient, drainage discharge, and sediment input. Sediment input can vary dramatically in response to factors such as fire suppression, road construction, natural grazing variation, vegetation change, soil formation, and climatic changes resulting in altered stream morphologies. For example, fire suppression will increase forest density, affecting runoff and reducing sediment supply, ultimately leading to incision. Changes in the migration patterns of herbivores can affect vegetation density as well. Additionally, climate changes can produce precipitation variation that results in runoff fluctuations and changes in vegetation patterns.

To gain a full understanding of the natural dynamics and limits of a system in resilient quasi-equilibrium, a more complete survey of the regional geomorphic response to climatic and anthropogenic variables should be considered. As a result, alternative efforts at restoration, including passive restoration techniques, can then be considered.

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NONNATIVE, PREDATORY FISH REMOVAL AND NATIVE FISH RESPONSE, UPPER VERDE RIVER, ARIZONA: PRELIMINARY RESULTS

John N. Rinne*

The estimated native fish fauna of the upper Verde River in Arizona has declined dramatically since 1994 (Rinne et al. 1998; Rinne 1999; Rinne et al. 2001; Rinne in press). By contrast, introduced, nonnative fish species have increased proportionally. Factors such as lack of flooding (Stefferud and Rinne 1995; Rinne and Stefferud 1997) and change in grazing management (Rinne 1999; Medina and Rinne 1999) have been suggested to be responsible, in part, for these recorded changes in fish community structure. However, the presence of nonnative fishes is perhaps the most direct, negative impact on the native species (Minckley 1973; Rinne and Minckley 1991; Minckley and Deacon 1991; Rinne and Stefferud 1997) in the upper Verde.

Because of the possible impact on native fish species through predation (Rinne 1995; Rinne and Alexander 1995) by large-sized (>100 mm) nonnative species such as smallmouth bass (*Micropterus dolomieui*), green sunfish (*Lepomis cyanellus*), and yellow bullhead (*Ameiurus natalis*), a pilot study was designed to remove these species from reaches of the stream and to determine the response by three of the long-lived, larger-sized native species: desert sucker, *Catostomus clarki;* Sonora sucker, *Catostomus insignis;* and roundtail chub, *Gila robusta*. The primary objective of the study was to remove predatory, nonnative species of fish from three approximately 1 km reaches in the upper Verde River. Change in recruitment of young-of-year (YOY) of the three native species between initiation of treatment and a year later was used as an indicator of response.

Study Area

Three 1 km treatment reaches were established in the headwaters of the upper Verde River (Figure 1). Two reaches were in the area of the established Burnt Ranch long-term monitoring site and one was below the 638 Road established monitoring site (Stefferud and Rinne 1995).

Methods

Fishes were sampled with backpack DC electrofishing gear. Reaches were sampled from down to upstream, normally with two units operating simultaneously. Because of instream and streambank vegetation that provided abundant cover (especially for smallmouth bass and green sunfish), electrofishing units were deployed in parallel along stream margins. Fishes were captured with dip nets by at least two individuals attending each unit. An additional two persons with dip nets followed 3–5 m downstream to increase the efficiency of capture for fishes that were missed when stunned and transported downstream by water current.

All fish were enumerated. Adult native species were counted and returned alive immediately to the water downstream of the electrofishing field; YOY were measured and similarly released. Nonnative fishes were measured, their stomachs were examined for food habits, and they were disposed of at the site.

Removal was performed initially in the autumn (October-November) of 1999 at all three experimental reaches, in the summer (June-July) at the two Burnt Ranch removal sites, and again in October of 2000 at all sites. Predator removal was conducted only twice, October of 1999 and 2000, at the 638 Road experimental removal reach. Pretreatment data consisted of initial samples (October 1999) at each site and independent samples at both Burnt Ranch and 638 Road in October 2000. In autumn of 2000, control reaches were sampled contiguous to the respective experimental reaches at Burnt Ranch and 638 Road. These reference reaches were used to document any annual, natural changes in YOY recruitment for the two suckers

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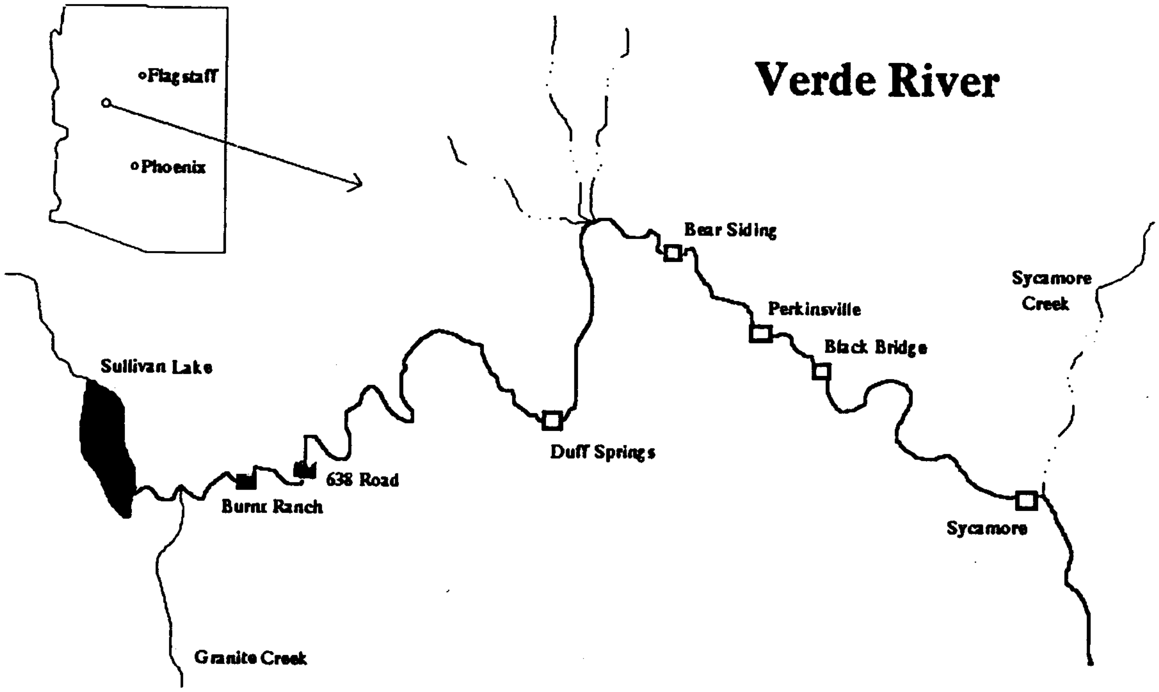

Figure 1. Map of upper Verde indicating the Burnt Ranch and 638 Road established monitoring sites where experimental removal was conducted.

and roundtail chub between autumn 1999 and 2000.

Results

Total numbers of fish captured in the experimental reaches in 1999 and 2000 are shown in Figure 2. Smallmouth bass was the dominant nonnative species present, followed by nearly equal numbers of green sunfish and yellow bullhead. Desert sucker dominated the native fishes captured, followed by Sonora sucker; roundtail chub were the least abundant. Bass were more abundant at the Burnt Ranch sites and averaged ca. 400 individuals per kilometer along the three experimental reaches. Roundtail chub increased successively downstream from the Burnt Ranch I to 638 Road reaches (Figures 1 and 2).

Through removal, smallmouth bass was reduced almost five-fold at Burnt Ranch I between autumn 1999 and July 2000; its abundance during the final removal activity in October 2000 was similar to July (Figure 3a). By comparison, no sunfish were captured during the initial removal at this site, but more than 80 individuals were captured in summer 2000 before dropping three-fold by the final removal event. Yellow bullhead were reduced five-fold from initial removal to summer 2000 but increased again by October 2000. The autumn 2000 reference sample indicated bass numbers similar to those in autumn 1999, but most were YOY individuals. Sunfish and bullhead numbers in the control sample, however, were lower than in all removal sample events.

At Burnt Ranch II, bass were reduced by only 50 percent by the final removal exercise in autumn 2000 (Figure 3b). Similar to Burnt Ranch I, sunfish increased in summer 2000 and declined almost three-fold by the autumn 2000 sampling. Bullheads followed a pattern similar to sunfish.

Recruitment of the native fishes responded to nonnative, piscine predator removal. YOY desert and Sonora sucker increased between autumn 1999 to 2000 at all removal sites (Figure 4). The numbers of YOY in control samples were almost identical between autumn 1999 and 2000. In contrast, no YOY roundtail chub were captured at the two Burnt Ranch sites, but 18 were collected at the 638 Road site in autumn 2000.

Discussion

Sampling during removal to reduce potential large-sized, nonnative predatory fishes at the three sites in the upper Verde indicated substantial populations of smallmouth bass. The one-year

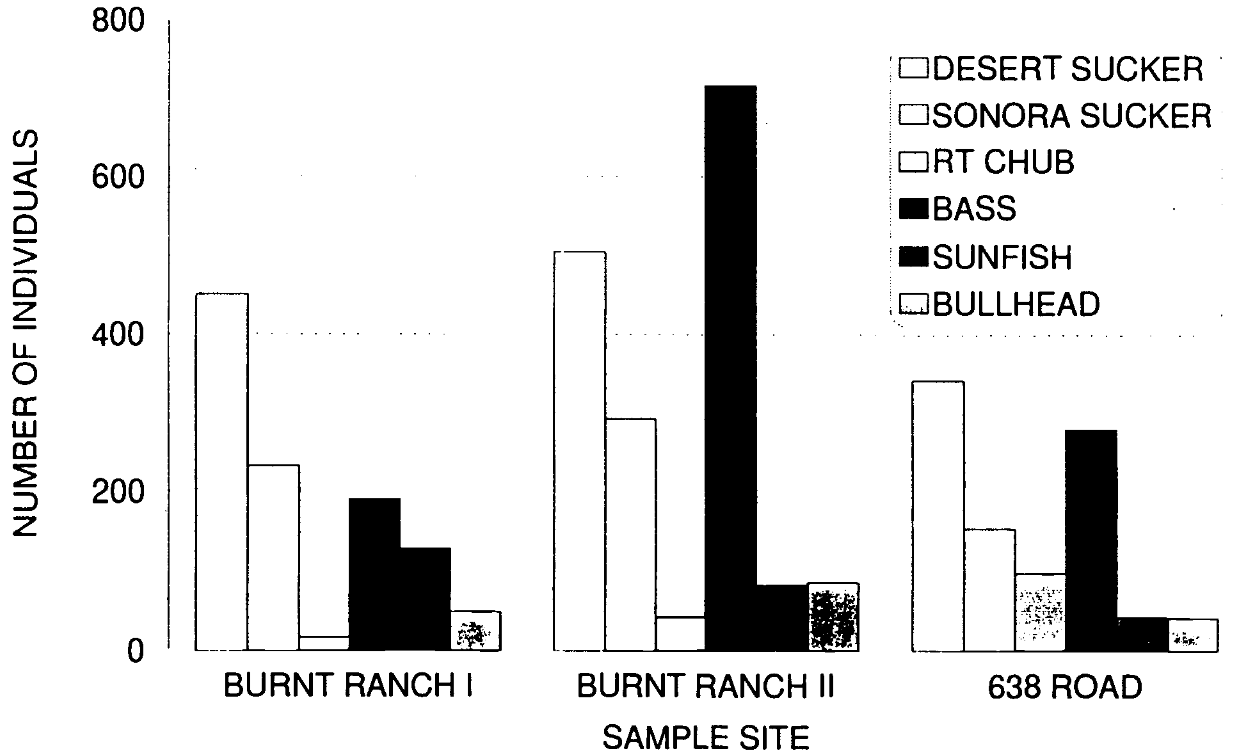

Figure 2. Total numbers of native and nonnative species captured during October 1999 to October 2000 at the three removal sites.

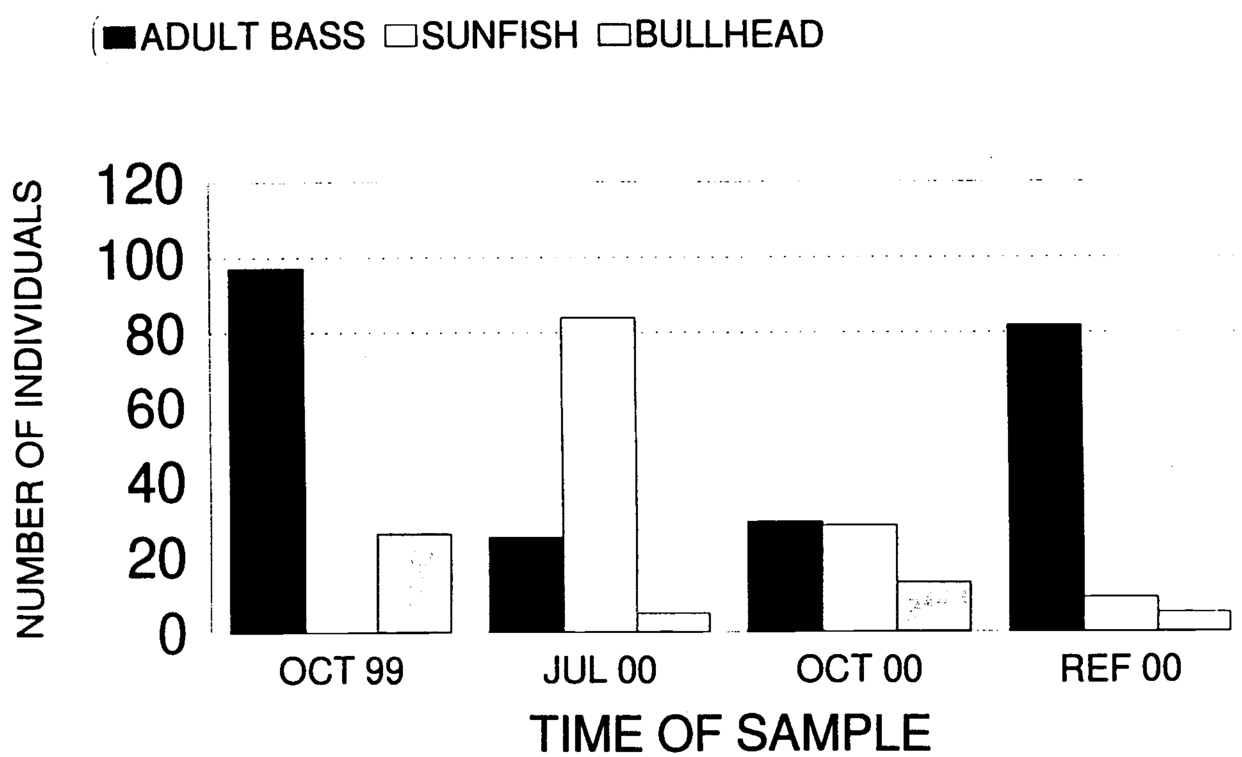

Figure 3a. Number of nonnative species removed over time at the Burnt Ranch I experimental reaches.

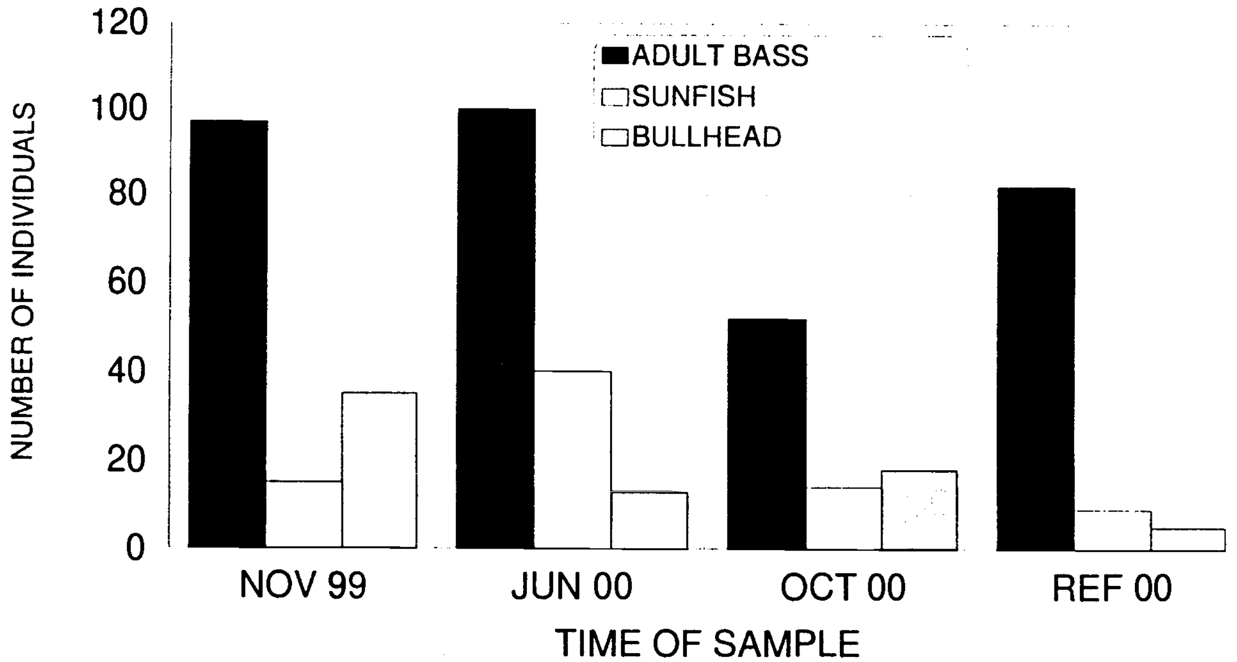

Figure 3b. Number of nonnative species removed over time at the Burnt Ranch II experimental reaches.

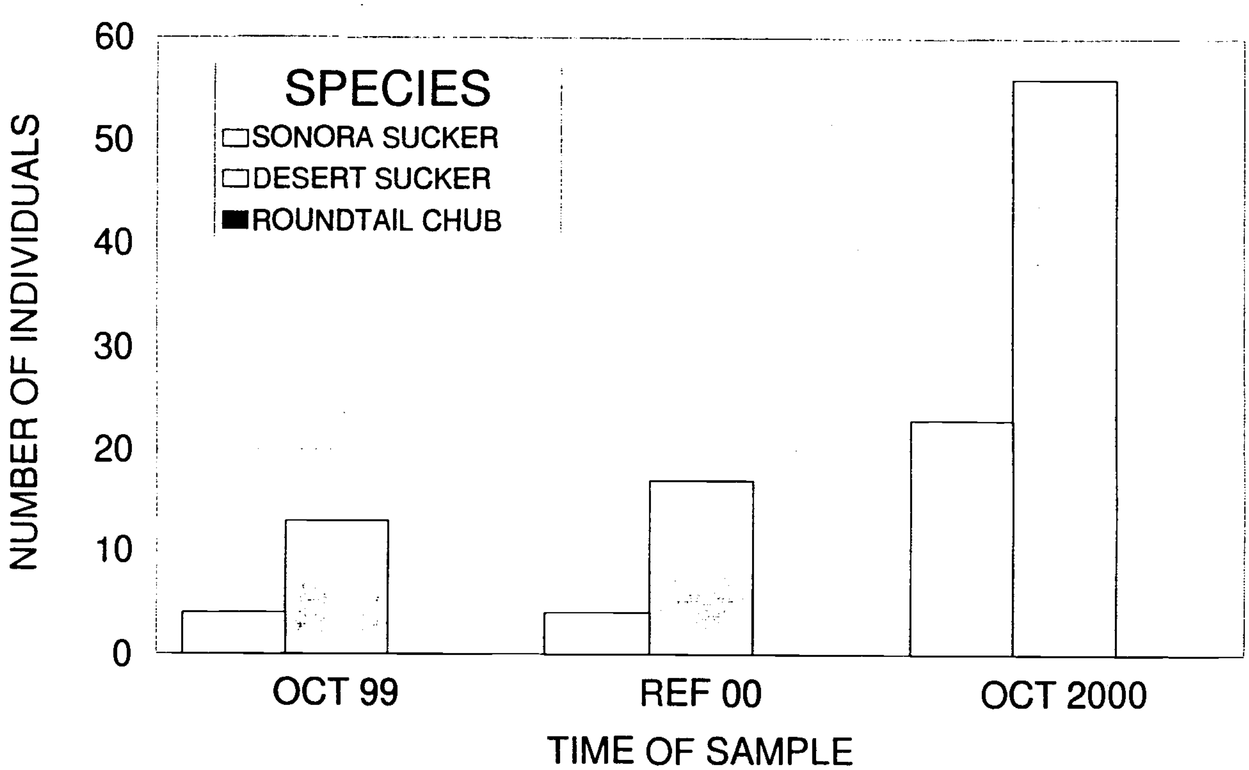

Figure 4a. Recruitment response of young-of-year native species with removal of nonnative species at Burnt Ranch I between autumn 1999 and autumn 2000.

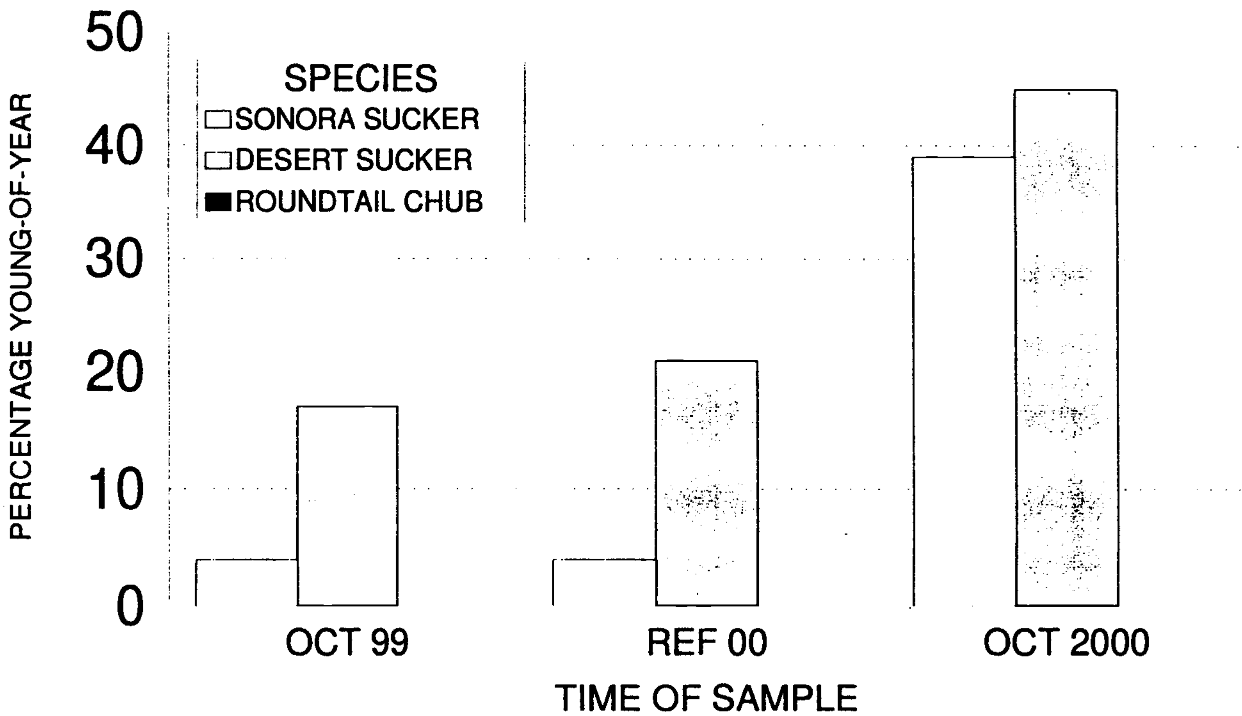

Figure 4b. Recruitment response of young-of-year native species with removal of nonnative species at Burnt Ranch II between autumn 1999 and autumn 2000.

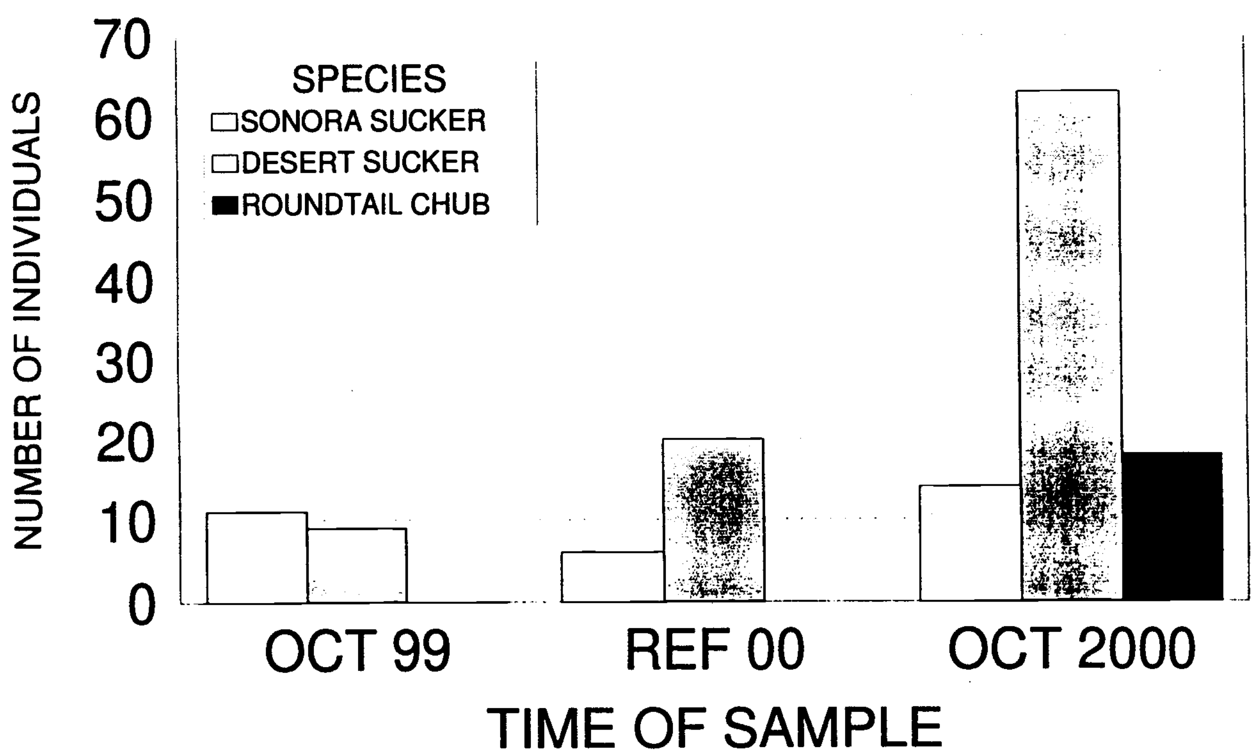

Figure 4c. Recruitment response of young-of-year native species with removal of nonnative species at 638 Road between autumn 1999 and autumn 2000.

pilot removal study suggests that predation is very likely a causative factor in the sustained reduction in populations of desert and Sonora sucker and roundtail chub. YOY of the two sucker species increased two- to three-fold between autumn 1999 and 2000 at all sites and roundtail chub appeared in samples at the 638 Road experimental removal reach.

Based on these results, nonnative, piscine predation also is very likely largely responsible for the almost total loss of the three smaller-sized native species, longfin and speckled dace and the threatened spikedace (Rinne et al. 1998). Only one longfin dace, no spikedace, and seven speckled dace were collected at established monitoring sites in the upper Verde River in 2000. In 2001, of the three small-sized native species, only two longfin dace were captured (Rinne in press). Because the adult size of these three species (50–80 mm) is similar to the size of YOY of the larger three species, and the response in recruitment of the latter group of native species, spikedace, longfin, and speckled dace are vulnerable to predation at all life stages. In contrast, by their second year of life (130+ mm), the three large-sized, native species may become less vulnerable to predation by the three nonnative fish predators in the upper Verde.

Longevity and reduction of predation vulnerability of the suckers and chub have permitted these three species to persist in greater numbers than the three small-sized species (Rinne in press). Habitat may also influence the relative abundance of the three large-sized species. The chub is an obligate pool dweller (Rinne and Stefferud 1996) and is often captured in association with cover. This native predator responded least to predator removal. Smallmouth bass also occupy pool habitats and prefer cover in southwestern rivers (Rinne et al. 2001). Chub increased successively downstream at the three removal sites (autumn 1999 sampling). The only positive response in this native species was at the lowermost removal site, 638 Road, where the species was most abundant initially in autumn 1999. Sonora sucker, also a pool dweller (Rinne and Stefferud 1996), responded less to predator reduction than did desert sucker. Desert sucker most frequently inhabits low to high gradient riffle habitats, especially at the YOY life stage, which conceivably reduced vulnerability to predation by bass. This native species responded most positively in all three experimental removal reaches (Figure 4).

Reference (control) reach sampling in autumn 2000 documented that there was not an innate, natural, annual increase in recruitment of the three natives. Nevertheless, study results are based on a single year of sampling and have to be considered preliminary. Another or even two additional annual replicates may be necessary before one can unequivocally state that nonnative, piscine predation is a primary, negative impact to the native fish fauna in the upper Verde River. Further, instream physical habitat changes resulting from a lack of significant (i.e. > 10 yr recurrence) flooding since 1995 and changes in grazing management are extant in the study area, as well as the influences of other introduced biological organisms, such as crayfish. These interactive factors also potentially affect native fish populations and must ultimately be factored into conclusions and future management of the native fish resource.

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BASE FLOW TRENDS IN THE UPPER VERDE RIVER REVISITED

Daniel G. Neary and John N. Rinne*

The native fish populations in Arizona's rivers are affected by various intrinsic and extrinsic factors that are characteristic of the arid environment of the Southwest and the activities of its human inhabitants (Table 1). Concerns have been raised about nearly all of these factors. However, climatic events of the past decade have focused questions on the potentially negative effects of reduced stream flows. Principal among these concerns are the effects of drought, consumptive water use for municipalities, water diversion for irrigation, and water quality deterioration on native fish. Minimum base flows are the most critical for fish survival. Significantly reduced base flows could put native fish populations, especially threatened and endangered species, at risk. Because of the controversy over threatened and endangered fish such as the spikedace (*Meda fulgida*) in the upper Verde River, it is important to examine the recent trends in minimum base flows on this river.

Because of the vagaries of climate, characteristic to the Southwest, and its aridity, patterns of stream flow are highly variable (Jaeger 1957; Green and Sellers 1964; Nations and Stump 1981). An alternating pattern of episodic floods and drought are the norm in the basin. The interactions of these factors result in the definition of aquatic habitat or the lack of it (Rinne 1995a). The upper Verde River is one of the few remaining reaches of wild, free-flowing rivers in Arizona. Furthermore, this reach of river is a rarity in the Southwest because it supports a native fish community (Stefferud and Rinne 1995). Study of the reach of river upstream from Sycamore Creek began in 1994 following major flooding events on the Verde in winter 1992–93. The main objective of the study is to examine the sustainability of the native fish fauna relative to abiotic and biotic factors. Specifically, the role of the changing hydrograph with time and space, and the effect of introduced fishes on the sustainability of the native fish fauna are being examined (Stefferud and Rinne 1995; Rinne and Stefferud 1996). Southwestern fishes are highly adapted to the varying cycles of feast (flood) and famine (drought; Minckley 1973; Deacon and Minckley 1974; Rinne and Minckley 1991). However, they are not able to adapt to the complete loss of surface flow, the obvious, critical component to their survival (Rinne 1995b).

Examination of the base flow hydrology in the upper Verde River basin was undertaken in 1997 to determine the trends in minimum base flows over the last 3 decades of the twentieth century (1963 to 1995; Neary and Rinne 1997). At the USGS Paulden gage, near the beginning of perennial flow on the Verde River, Neary and Rinne (1997) found that mean daily minimums ranged from 0.42 to 0.71 m^3/sec (15 to 25 ft^3/sec). The annual minimum mean daily flows at this gage exhibited an increasing trend through this period. At the Clarkdale gaging station above Cottonwood, mean daily minimums varied from 1.70 to 2.32 m^3/sec (60 to 82 ft^3/sec). Similar to Paulden, the trend at Clarkdale since 1965 was toward increasing annual minimum mean daily flows. At the time of our initial analysis, we concluded that the increases in minimum base flows on the upper Verde River appeared to indicate that adequate flows would be available in the near future to sustain the Verde's fish population (Neary and Rinne 1997). However, we cautioned that the base flow trend would require future evaluation and monitoring to determine if rapid urbanization of the Prescott and Chino Valley areas in future years could impact the base flow of the upper Verde River.

Because of recent droughts and the extensive, rapidly increasing urban development in Chino Valley, which is the headwater of the Verde River drainage system, concerns have arisen again over the ability of the river to sustain its native fish population. An understanding of this trend is very

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Table 1. Ecosystem influences on native fish in the Upper Verde River watershed.

Uplands Extrinsic – Natural	River Intrinsic	Uplands Extrinsic – Human Related
1. Climate	1. Stream flow	1. Grazing
2. Geology	2. Nonnative fish	2. Mining
3. Soils	3. Other nonnative fauna	3. Roads
4. Vegetation	4. Water quality	4. Forestry
5. Groundwater	5. Geomorphology	5. Urban development
6. Runoff	6. Sediment regime	6. Agriculture
7. Natural sediment	7. Macro-invertebrates	7. Recreation
8. Wildlife	8. Riparian vegetation	8. Water engineering

important for resource managers to arrive at decisions regarding land use activities and sustainability of the Verde River spikedace (*Meda fulgida*) and other native fish. This paper is an update of the 1997 Neary and Rinne paper. It reexamines base flow trends in the upper Verde River in light of recent trends in urban development in Chino Valley and Prescott to determine potential impacts on the habitat and sustainability of the native fish fauna. This paper deals only with data from the Paulden stream gage in the uppermost portion of the Verde River.

Methods

Stream flow data were obtained from the U.S. Geological Survey, Water Resources Division, Arizona District, records for the Paulden stream gage (No. 9503700) from 1963 through 1999. Although Neary and Rinne (1997) discussed data from the Clarkdale gage (No. 9504000) and the Camp Verde gage (No. 9506000), this analysis considers only records from the Paulden gage. The Paulden gage is located on the upper Verde River (perennial flow) between the confluence of Granite Creek (ephemeral flow) and Hell Canyon (ephemeral flow), about 14 km below the beginning of perennial stream flow. Its contributing drainage is about 5,568 km^2, or 40 percent of the Verde's 14,000 km^2 basin area (House et al. 1995). Flow data are available on the Verde Watershed Association's Web site at <http://www.verde.org>. Monthly and annual discharge data were taken from U.S. Geological Survey gage summary tables. Mean daily flows were analyzed to determine the minimum mean daily flow on an annual basis. Precipitation data were taken from the Prescott precipitation station (No. 026796) for the period 1931 through 1999; the rain gage is about 46 km south of the Paulden gage.

Results and Discussion

Base Flow Trends

Mean monthly discharges at Paulden from 1964 to 1999 ranged from 0.65 to 3.62 m^3/s (23 to 128 ft^3/s). The mean maximum discharge of 40.77 m^3/s (1,440 ft^3/s) normally occurs in February, and the mean minimum discharge of 0.45 m^3/s (16 ft^3/s) occurs in May. During the period of record (1964–1999), the maximum peak daily flow was 387.85 m^3/s (13,700 ft^3/s) in 1993. An instantaneous peak flow of 657 m^3/s (23,200 ft^3/s) occurred during the 20 February 1993 storm (House et al. 1995). The minimum daily flow was 0.42 m^3/s (15 ft^3/s) during an 11-day period in 1964.

The annual minimum daily flows for the period of record, representing the lowest base flow during each year, are shown in Figure 1. They range from a low of 0.42 m^3/s (15 ft^3/s) in 1964 to a high of 0.71 m^3/s (25 ft^3/s) in 1985, 1986, 1994, and 1995. There is a trend of increasing annual minimum daily flows over this period. The cause of the slight downturn since 1995 may be nothing more than natural, climate-related oscillations observed in other years. It is certainly within the measured range of variability. The 5-year running average rose from 0.52 m^3/s (18.4 ft^3/s) in 1967 to 0.68 m^3/s (24.2 ft^3/s) in 1995, a 32 percent increase. As of 1999, there does not appear to be any evidence to indicate that water use in the upper Verde River watershed over the past 3 decades has affected the annual minimum daily flow.

Long-Term Trends

Stream flow gaging at Paulden did not start until 1963, so it is difficult to determine where the current trends in minimum daily base flow fit in the long-term pattern of minimum daily base flows. A reconstruction of climate over the past 2200 years by Grissino-Mayer (1996) indicates that

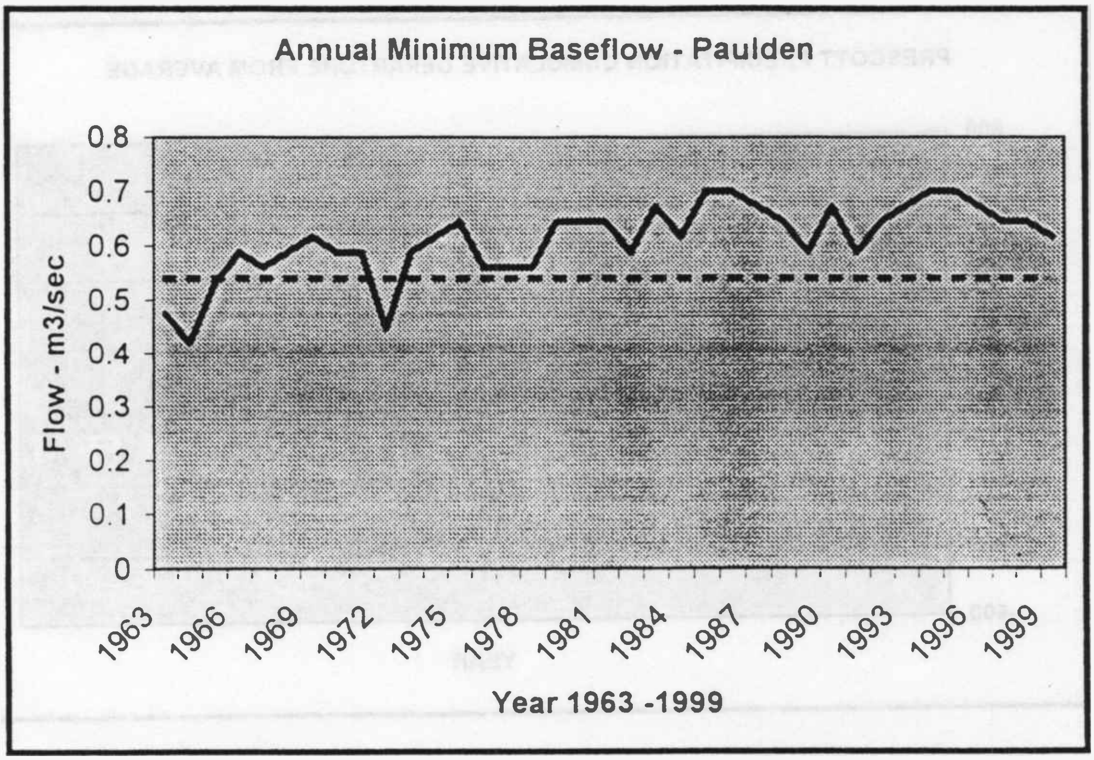

Figure 1. Annual minimum base flows, Paulden gage, upper Verde River, 1964–1999.

the current conditions are the wettest since 600 A.D.

Mechanisms

The trend of increasing annual minimum daily flows at both the Paulden and Clarkdale stream gaging sites suggests that base flows in the upper Verde River watershed are increasing in response to increases in precipitation. There is no evidence of decreases in evapotranspirational losses. No major vegetation management programs have occurred in the past 3 decades that would reduce transpirational losses of water. Photo comparisons show that the opposite trend may be occurring due to increasing pinyon-juniper density and biomass. The remaining mechanisms that might explain the increases in minimum daily base flows are reduced groundwater pumping for agriculture in the Chino Valley and increased precipitation.

Agriculture Groundwater Pumping

Wirt and Hjalmarson (2000) reported a decline in agricultural usage of groundwater in Big Chino Valley from a peak of just over 14.8 million m^3/yr (12,000 ac-ft/yr) in the mid 1970s to 2.5 million m^3/yr (2,000 ac-ft/yr). This reduction coincided with an increase in the mean annual base flow of the Verde River and an increase in winter groundwater levels in Big Chino Valley. Wirt and Hjalmarson (2000) concluded that this is additional evidence of the direct linkage between Big Chino Valley groundwater and base flows in the Upper Verde River.

Rainfall

Although Wirt and Hjalmarson (2000) discounted precipitation as the cause of increased base flow in the Verde, during the 1932 to 1999 time frame there have been wetter and drier periods. These are indicated by cumulative departure curves (e.g. wetter, 1932–1946; drier, 1946–1964; wetter, 1965–1972; drier, 1973–1981; wetter, 1981–1988; drier, 1989–1999). In Figure 2, the 1964–1999 segment shows a general wetting trend during the period of increasing minimum daily flows in that cumulative departures from the mean have be-

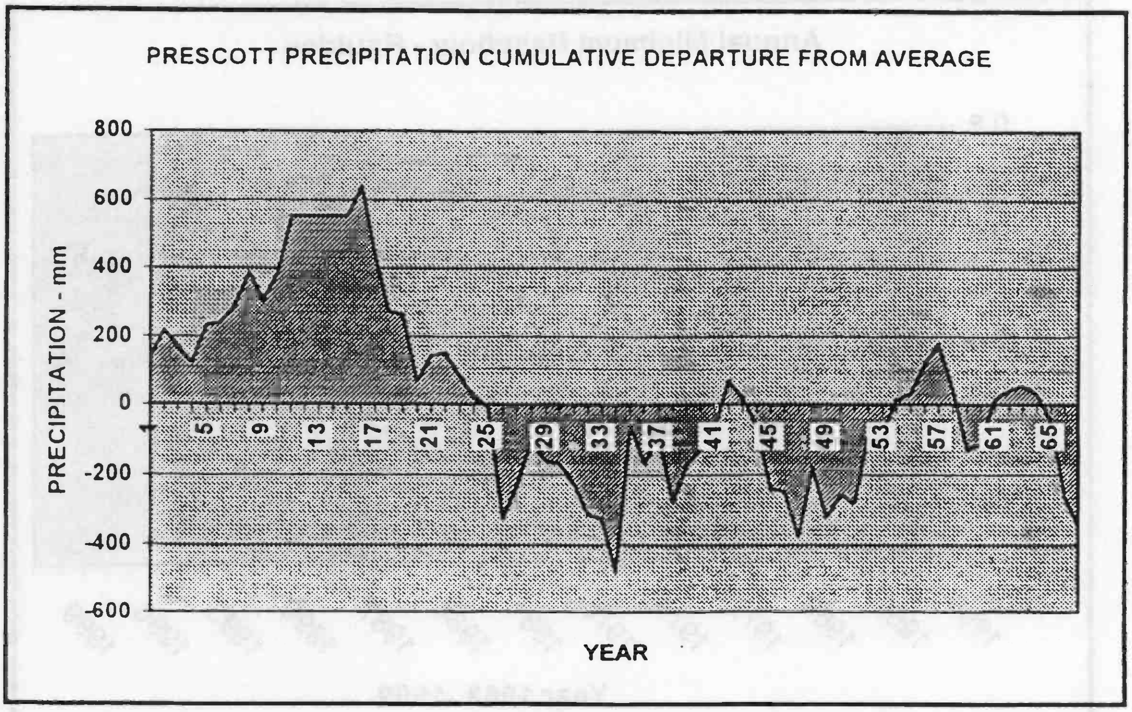

Figure 2. Prescott precipitation cumulative departure from average, 1932–1997.

come more positive since the low point in 1964. Another indication that there is a trend since 1964 toward wetter weather is shown in Figure 3. The number of days with base flows at or below the mean minimum flow has declined dramatically since the 1960s and 1970s. This trend could be attributed to the decline in agricultural groundwater pumping. However, the storm flow trend during this period supports the wetting trend hypothesis. This information suggests that annual minimum daily flows might respond to oscillations in precipitation and not some other hydrologic process such as evapotranspiration.

A downward trend in vegetation density in the watershed could possibly reduce evapotranspiration and increase base flows. Based on photographic evidence, vegetative cover in the upper Verde River riparian zone over the past 30 years has increased, not decreased. There is not enough evidence of vegetation cover expansion on upland areas in the past 3 decades to argue one way or the other, but vegetative cover since the early part of the twentieth century has definitely increased.

Storm Flow

During the period of time in which annual minimum daily base flows at the Paulden gage have been increasing, annual maximum daily peak flows have also been increasing (Figure 4). Because peak flows have a high degree of year-to-year variability, they are indicators of an increasingly wetter climate. Peak flows have definitely increased over the period of record, culminating in the high flows of February 1993, which were estimated to have a return period of 70 years.

Potential Urban Groundwater Pumping

A recent proposal by the city of Prescott to pump up to 17.0 million m^3 (45 billion gallons) of groundwater from Big Chino Basin could seriously impact minimum daily flows on the Verde. Pumping the full allotment (equivalent to 0.54 m^3/sec; dotted line in Figure 1) could significantly affect base flow in the upper Verde River in the driest of the past 38 years. Wirt and Hjalmarson (2000) concluded that 80 percent or more of the upper Verde's base flow comes from interconnected

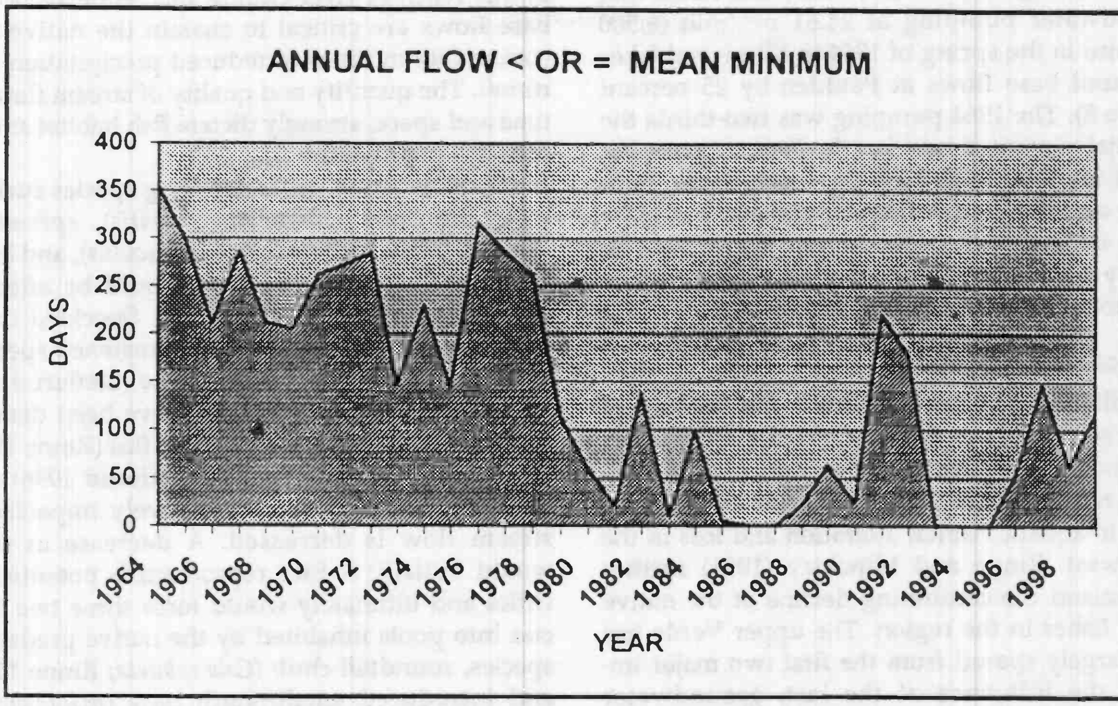

Figure 3. Number of days with base flow less than or equal to the mean minimum, Paulden Gage, 1964–1999.

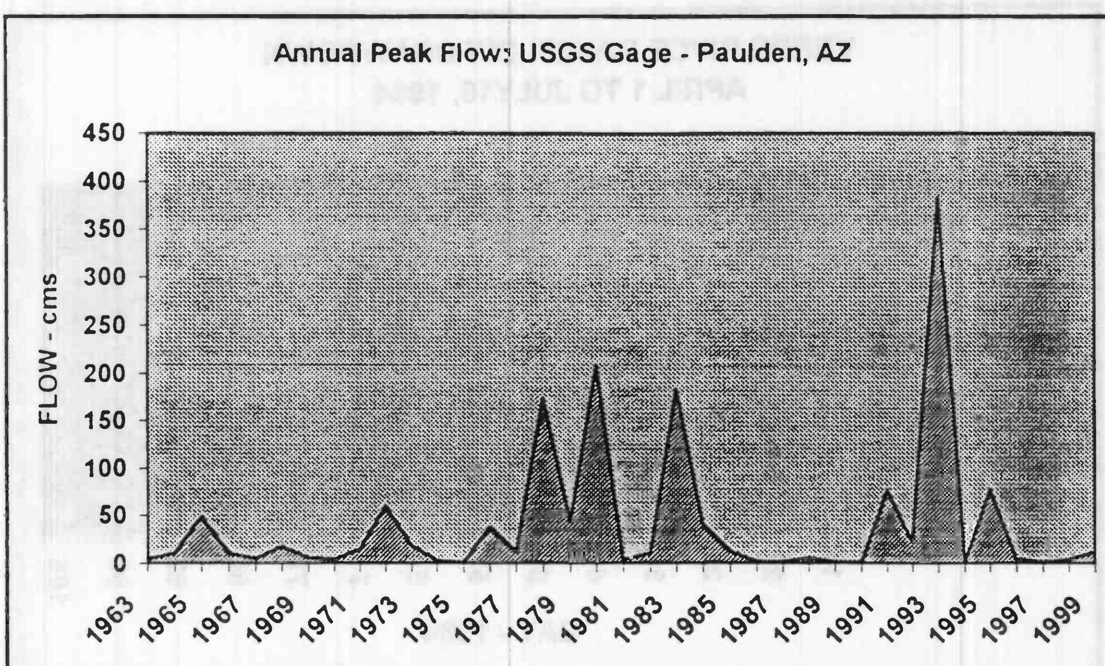

Figure 4. Annual peak flow at the upper Verde River stream gage at Paulden, 1964–1999.

aquifers in Big Chino Valley. They also noted that groundwater pumping at 24.61 m^3/min (6,500 gal/min) in the spring of 1964 to fill several lakes decreased base flows at Paulden by 25 percent (Figure 5). The 1964 pumping was two-thirds the potential maximum rate that the Prescott pumping would involve. With base flow reductions, both native and non-native fish populations would be forced into remnant pools, thereby aggravating an already serious predation problem that is contributing to the decline of native fish species.

Significance to Fishes

Miller (1961) first reported the decline of native southwestern fishes as a result of man's activities. Dam construction, diversion, and groundwater mining were listed as major factors that result in aquatic habitat alteration and loss in the Southwest. Rinne and Minckley (1991) further emphasized the continuing decline of the native desert fishes in the region. The upper Verde has been largely spared from the first two major impacts; the influence of the last, groundwater pumping, may be yet to come. Just as floods have been documented to be beneficial to native fishes in this reach of river (Rinne and Stefferud 1996), base flows are critical to sustain the native fish community in times of reduced precipitation and runoff. The quantity and quality of stream flow, in time and space, strongly dictate fish habitat and in turn fish populations.

Shallow water, riffle-dwelling species such as loach minnow (*Rhinichthys cobitis*), spikedace (*Meda fulgida*), speckled dace (*R. osculus*), and longfin dace (*Agosia chyrsogaster*) would be affected first if stream flow is decreased. Speckled dace, longfin dace, and spikedace, a threatened species, currently inhabit the upper Verde (Stefferud and Rinne 1995). All three species have been demonstrated to inhabit low-gradient riffles (Rinne 1992; Neary et al. 1996; Rinne and Stefferud 1996) and would be the first to be negatively impacted if stream flow is decreased. A decrease in flow would initially reduce reproductive potential in riffles and ultimately would force these two species into pools inhabited by the native predatory species, roundtail chub (*Gila robusta*; Rinne 1992) and introduced smallmouth bass (*Micropterus dolomieui*) and yellow bullhead (*Ictalurus nebulosus*; Rinne and Neary 1997).

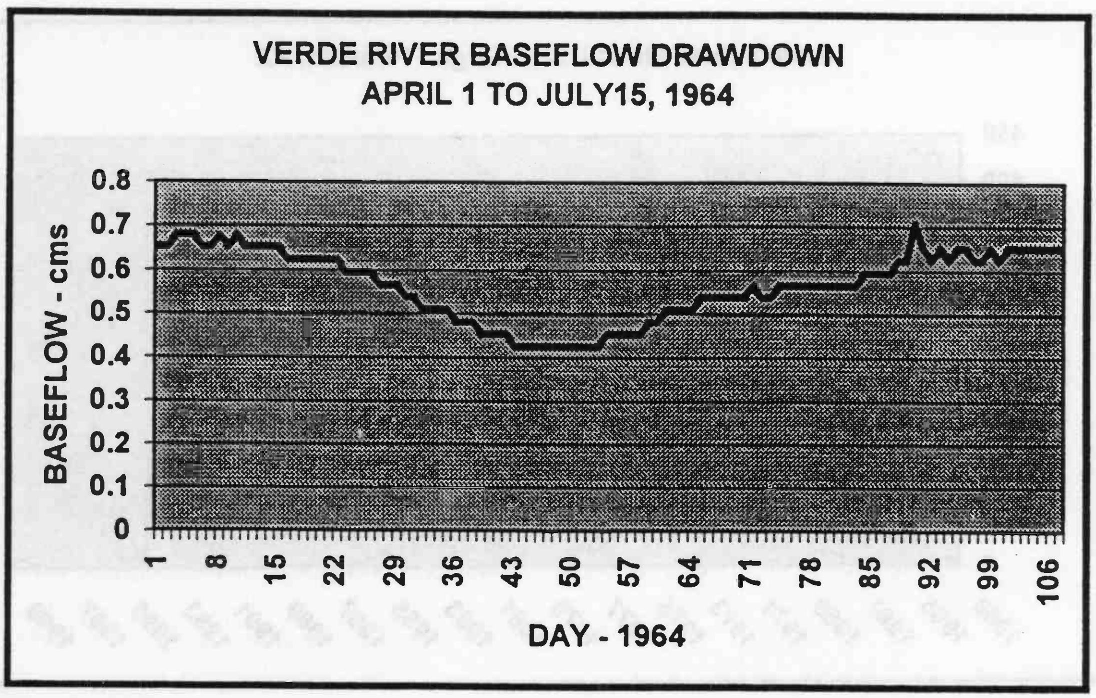

Figure 5. Base flow drawdown from groundwater pumping in the Chino Valley, Paulden Gage, April 1–July 15, 1964.

Conclusions and Recommendations

At the USGS Paulden gage, near the beginning of perennial flow on the Verde River, mean daily minimums range from 0.42 to 0.71 m^3/sec (15 to 25 ft^3/sec). The annual minimum mean daily flows at this gage have exhibited an increasing trend over the past 3 decades. At the present time, these increases in minimum base flows on the upper Verde River appear to indicate that current flows are adequate to sustain the Verde's native fish population. It is evident from new information developed by the U.S. Geological Survey that base flows in the upper Verde River depend heavily on groundwater in the Big Chino aquifer. The current trend in minimum base flows will require close evaluation and monitoring in the future. Preliminary information on potential groundwater withdrawals by the city of Prescott and the rapidly urbanizing Chino Valley indicate that these actions could begin to impact the base flow of the upper Verde River.

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THE RESPONSE OF SALT CEDAR, *TAMARIX CHINENSIS*, TO EXPERIMENTAL FLOWS IN THE GRAND CANYON

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Abstract

We examined the response of an exotic tree, *Tamarix chinensis*, to an experimental low steady flow and a four-day spike flow along the Colorado River in the Grand Canyon. Plant density data in newly exposed beach areas were collected for 52 transect lines throughout the river corridor study site. These data were collected over a 3-month period at a time when the river flow remained near 8,000 cfs and once after a spike flow of approximately 31,500 cfs, plant capacity flow. During the experiment a significant establishment of *Tamarix* seedlings was observed in the newly exposed beach areas, but a 31,500 cfs spike flow in the fall resulted in a similarly significant mortality in the seedlings. The experimental low steady flow through the Grand Canyon was designed to create habitat and a food base for native endangered fish. It is important to understand the different responses that an experimental hydrograph will have on a system when making management decisions. From the results found in this study, it appears that the rapid establishment of exotic plant seedlings needs to be taken into consideration when altering a stream-flow regime.

Introduction

Riparian ecosystems are an endangered environment in the American Southwest, comprising less than 3 percent of the total landscape area (Naiman and Decamps 1997). Riparian plant communities have several effects on riverine ecosystem functions. Stabilization of stream channels (Riedl and Zachar 1984) and stored sediments (Lowrance et al. 1986) are attributed to riparian vegetation. In addition, riparian plants act as nutrient sinks and improve water quality for the surrounding watershed (Schlosser and Kar 1981). They influence water temperature through shading, decrease flood peaks by providing flow resistance, and act as important recharge points for restoring ground water supplies (Debano and Schmidt 1990; McGlothin et al. 1988). A wide variety of animals and plants are found within riparian habitats, and these areas are known for supporting a great amount of biodiversity (Carothers 1977). Therefore, river ecosystem processes and biodiversity are dependent on their surrounding riparian plant community.

River processes are drastically altered by dam construction. Dams alter river hydrology by reducing sediment load, causing channel incision, and drawing down water tables (Reily and Johnson 1982). Dam construction also promotes the establishment of exotic species (Smith et al. 1998) and alters the fauna that are dependent on historical river conditions. Dams are also known to decrease the temperature of water flowing through a river corridor by releasing cold water from the bottom of a reservoir or lake.

Tamarix is a summer-germinating species that covers approximately 500,000 ha of floodplain land in the western United States (Stromberg 1997); it is a dominant tree in the Grand Canyon. This exotic plant is responsible for disrupting native willow and cottonwood stands in many southwestern riparian areas (Ohmart and Anderson 1982). *Tamarix* was introduced to the United States during the nineteenth century as an ornamental plant (Robinson 1964). These trees are adapted to establish after spring flood disturbances (Brock 1994) in moist open sites (Everitt 1980). *Tamarix* can be reproductively mature the first year following establishment (Brock 1994), and can produce prolific amounts of seeds through much of the growing season (Warren and Turner 1975).

The Colorado River is one of the most used and anthropogenically manipulated rivers in the world (Malanson 1993). Since the construction of Glen Canyon Dam in 1963, the Colorado River corridor has been critically altered with regard to the disturbance regime, sediment load, flora, and

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many other biotic and abiotic factors (Webb et al. 1999). Experimental flows took place on the river from May through September of 2000. The purpose of the low steady flows was to create habitat and a food base for the native endangered humpback chub juveniles (*Gila cypha*). The stream flow hydrograph during this period consisted of a four-day spike flow of approximately 33,200 cfs in May and also in September; between the spike flows, water was maintained at a low steady flow of about 8,000 cfs. The spike flow in May was designed to allow juvenile fish to gain access to tributary mouths. After the spike flow the water in the Colorado River corridor was decreased to about 19,000 cfs to create a pooling effect. Managers hypothesized that the pooling would keep the juvenile fish from getting swept from tributary mouths into the main-stem Colorado River.

We know that hydrologic regimes greatly influence the structure and function of riparian vegetation (Bedford 1996; Poff et al. 1997). The purpose of our study was to observe *Tamarix chinensis* seedling colonization in the beach areas that are exposed by lowering the water in the river corridor from power plant capacity (31,500 cfs) to a low steady flow of 8,000 cfs. Here we examine the response of *Tamarix chinensis* over time to low steady flows and a September spike flow.

Methods

Our study sites are along the Colorado River corridor through Marble Canyon and Grand Canyon, in Arizona. Throughout the summer of 2000, water was released out of Glen Canyon Dam at an approximate steady flow of 8,000 cfs with two spike flows of 31,500 cfs in May and September. These experimental flows marked the first time in history that the Colorado River was being managed for biological purposes. The study area is dominated by small beaches composed of fine-grained sediment. Vegetation along these beaches consists of herbaceous riparian plants (*Equisetaceae*, *Juncaceae*, and *Cyperaceae*), exotic *Tamarix chinensis* trees, and some native willow (*Salix exigua*). Farther away from the river and up the canyon walls the vegetation shifts dramatically from riparian to arid plants such as *Agavaceae* and *Cactaceae* species.

Sites were selected based on the vegetation present and the structure of the beach. We selected beaches with few cobble bars and other obstacles that could potentially impede seedling establishment. Most of the sites are in reaches of the river corridor where sand bar studies are being conducted, so topographic layouts are readily available. Figure 1 shows the spatial distribution of the sites along the river. Sites RM1, RM7, and RM68

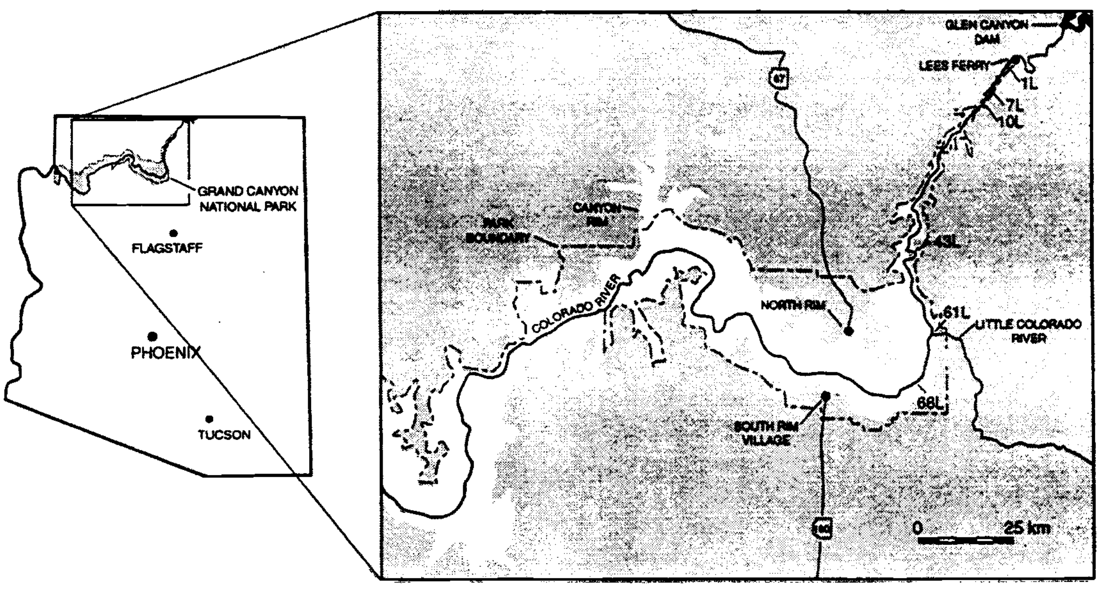

Figure 1. The location of the six sites surveyed for this study. Sites RM1 and RM7 are in close downstream proximity to the Paria River confluence. Site RM68 is downstream of the Little Colorado River. Sites RM11, RM43, and RM61 are not near any tributary confluences.

are downstream of confluences with the Paria and Little Colorado Rivers.

Vegetation density was measured at specified transects within a site. Vegetation was censused using a 0.25 m^2 at each meter down each transect. *Tamarix* seedlings in this area were counted in a 0.25 m^2 from the 19,000 cfs edge down to the approximate 8,000 cfs edge of water. These data were collected in June, July, and August, and in September after the four-day spike flow of 33,200 cfs. The data were analyzed using Manova repeated measures analysis of time in JMP 4 (SAS Institute). Dam discharge and flow data were collected from USGS gauges at Lees Ferry (RM0) and at Phantom Ranch (RKM124.5).

Results

The stream flow hydrograph taken at Lees Ferry shows the amount of water flowing out of Glen Canyon Dam from May through September 2000 (Figure 2). The dam released a 4-day spike flow in May followed by a pooling period and flow decrease to 8,000 cfs. There was very little variation in the water level for the remainder of the summer. The water remained low and steady at 8,000 cfs. At the beginning of September another four-day spike flow occurred, and water flowed out of the dam at 31,500 cfs.

Tamarix seedlings showed a time response between the June and August sample (Figure 3) when we saw a significant increase in tamarisk seedling growth ($F = 93.2556$, $P < 0.0001$). From June to July there was a 34 percent increase in seedling establishment, and from July to August another 68 percent increase occurred in *Tamarix* seedling establishment. After the September spike flow the presence of *Tamarix* seedlings decreased ($F = 22.4511$, $P < 0.0001$). The seedling count in September showed a 64 percent reduction from the August sample.

Different sites responded differently through time (Figure 4). Sites RM1, RM2, and RM68, which are closer to major confluences of the Paria or Little Colorado River, experienced significantly more tamarisk seedling growth from June through August ($F = 2.6272$, $P < 0.0074$) than sites RM10, RM43, and RM61. Sites RM1, RM2, and RM68 also experienced significantly ($F = 4.7321$, $P < 0.0014$) less tamarisk seedling mortality after the 4-day spike flow in September.

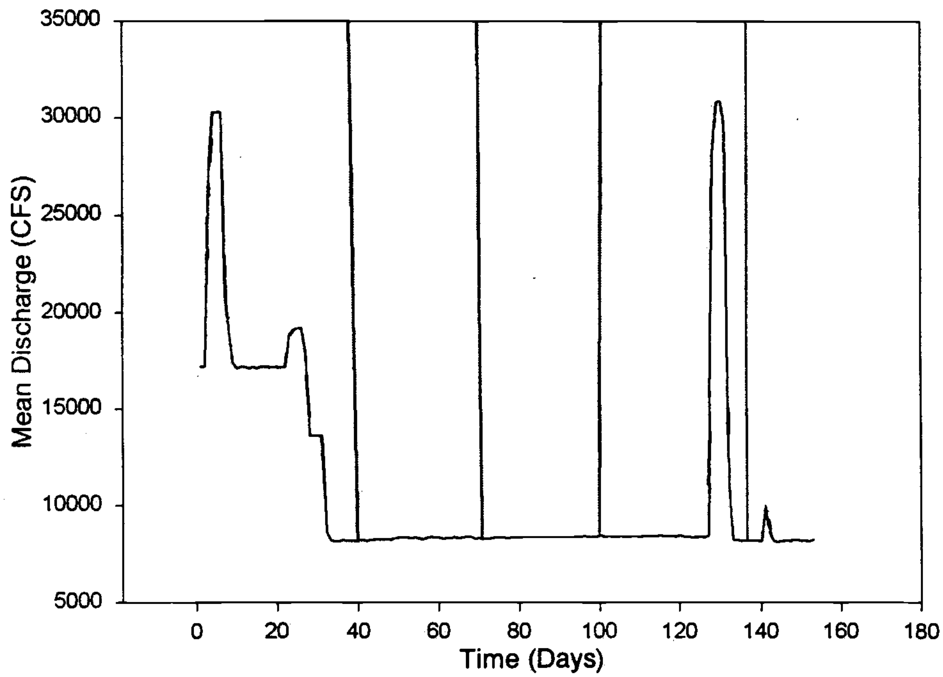

Figure 2. Mean discharge from USGS gauge at Lees Ferry, AZ (in cubic feet per second, cfs). The vertical lines indicate sampling dates throughout the summer's steady low flows. A spike flow went through the system at approximately day 30.

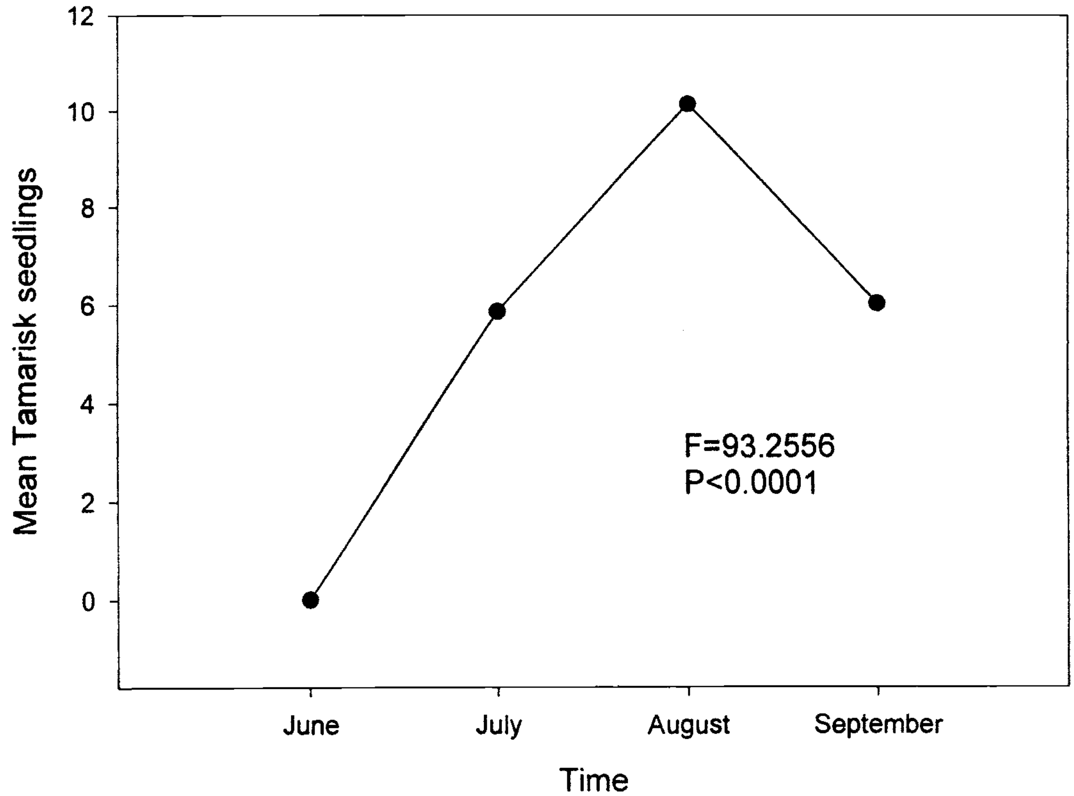

Figure 3. *Tamarix* seedling establishment, June to September. There was a 38% increase from June to July, a 68% increase from July to August, and a 64% reduction after the spike flow in September.

Because we determined a site by time interaction, we did an analysis on individual site differences over time. For sites RM1 and RM7 below the Paria confluence and for RM68 below the Little Colorado River confluence we did not determine a significant difference within sites over time ($F = 0.5861$, $P = 0.6392$; $F = 0.5333$, $P = 0.6739$; and $F = 2.0519$, $P = 0.2493$ respectively). Sites RM10, RM43, and RM61 without an upstream confluence had significant time differences within sites ($F = 0.0714$, $P = 0.9667$; $F = 0.0774$, $P = 0.9689$; and $F = 1.7634$, $P = 0.3264$ respectively).

Discussion

Al six sites showed significant tamarisk seedling establishment from June through August. However, following the 4-day spike flow in September, there was enormous seedling mortality; a 64 percent reduction in *Tamarix* seedling density was observed in September compared to the August sample. We attribute the seedling mortality to the spike flow. These results showing inundation mortality are consistent with the literature; it is well documented that salt cedar does not respond well to fall flooding and inundation (Gladin and Roelle 1998; Stromberg 1997, 1998).

This research on *Tamarix chinensis* has many implications for management of the Colorado River in the Grand Canyon. Rapid establishment of *Tamarix* seedlings along newly exposed beaches should be considered when managing for native endangered fishes. Converse et al. (1998) have shown that fish densities are nearly twice as great along vegetated shorelines as on talus or debris fans. Managers can therefore allow *Tamarix* to help create fish habitat, but this increases the range of this exotic tree, which dominates beaches in the Grand Canyon. Other studies (Stromberg 1998) have shown that *Tamarix* and cottonwood (*Populus fremontii*) are essentially equivalent in riparian ecosystems along free-flowing rivers. If this is the case along the regulated Colorado River, then perhaps *Tamarix*, although an exotic tree, can maintain the riparian functions of the system and provide fish habitat.

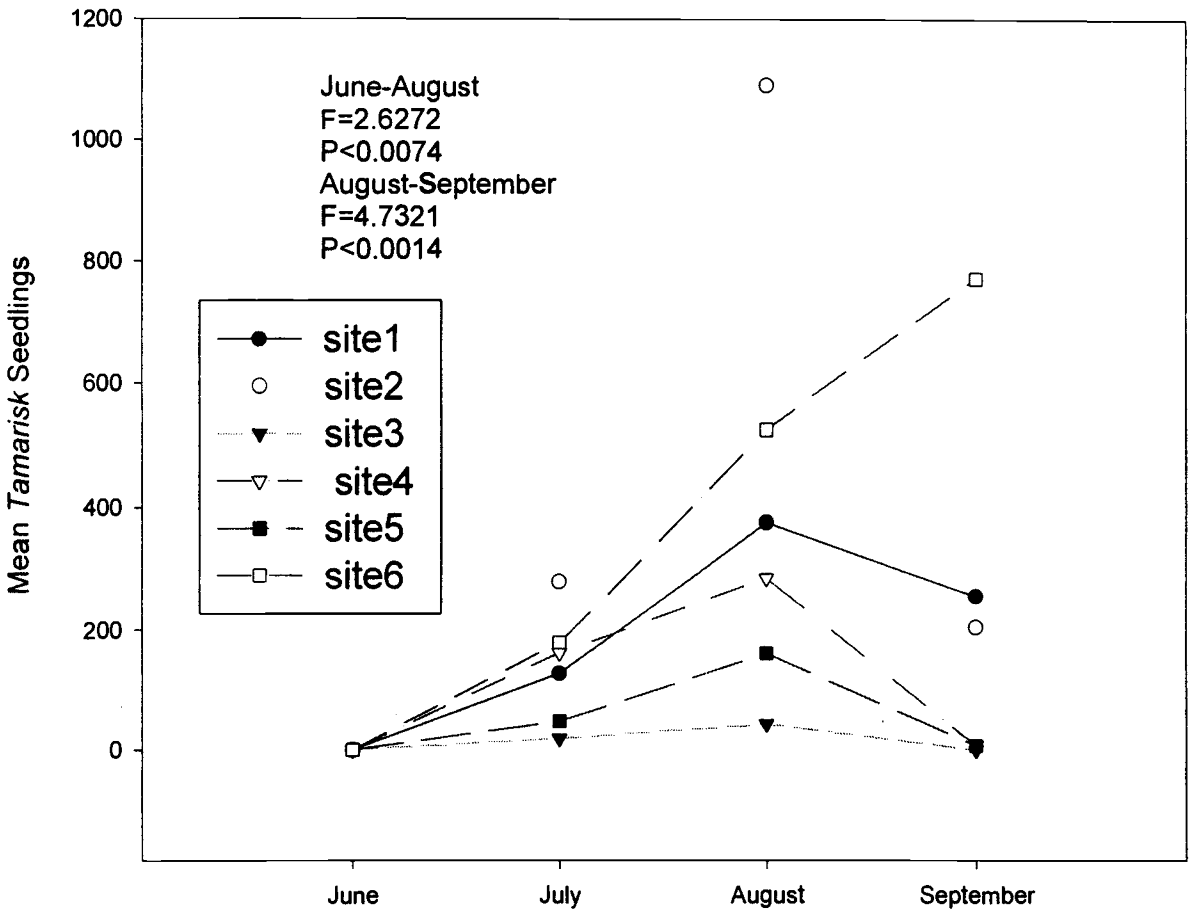

Figure 4. Establishment of *Tamarix* seedlings by site during steady low experimental flow at 8,000 cfs from June through August 2000. Seedlings at sites 1, 2, and 6 downstream from confluences display the most growth.

Geology can also affect the establishment of seedlings. Reaches of the river considered to be wide may have more solar input for better growth, and their shallower slopes allow vegetation to be closer to the water table and less likely to be affected by floods than reaches of the Colorado River considered to be narrow (Schmidt and Graf 1988). All of the sites in this paper are in reaches considered to be wide, with shallow slopes. Sites RM1, RM7, and RM68 have a greater average unit of stream power in pounds per foot than sites without confluence inputs. Sites RM11, RM43, and RM61 also have a greater number of re-circulation zones than sites with confluence inputs.

A final consideration is the quality of recreation in the Grand Canyon. Exposing new beaches, a prime recreation resource, to tamarisk establishment could potentially diminish the usable area of those beaches for camping as the seedlings grow larger. Ultimately, the encroachment of vegetation is almost as important as erosional processes in decreasing beach areas along the Colorado River corridor (Kearsley et al. 1999; Webb et al. 1999).

In summary, *Tamarix chinensis* established seedlings prolifically from June through August during the steady 8,000 cfs flows on the Colorado River through the Grand Canyon. It then experienced great mortality following a 4-day spike flow of 33,200 cfs (power plant capacity) in September. These data provide biology, hydrology, and recreation management implications for the area.

Acknowledgments

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PREDICTING THE VOLUMETRIC WATER CONTENT OF IRRIGATED ARIZONA SOILS AT DIFFERENT SOIL WATER POTENTIALS

Mohammed H. Bagour and Donald F. Post*

The volumetric water content (θ_v) was measured at −10, −33, −100, and −1500 kPa of soil-water potentials for 10 irrigated Arizona soils. The soils ranged in texture from sand to silty clay, and the soil properties of percent clay, percent sand, stickiness, and plasticity were measured by eight professional field soil scientists. A mean was calculated for these four properties for each soil and was regressed against θ_v for these 10 soils at the four different soil-water potentials. The bulk density was also measured, using both the absolute densities and a bulk density rating. The mean simple linear regression R^2 coefficients of determination were 0.89–0.91, 0.90–0.93, 0.75–0.81, 0.81–0.82, and 0.52–0.59 for percent clay, percent sand, stickiness, plasticity, and bulk density rating, respectively. Multiple linear regression equations were also computed and the mean R^2 was 0.98. The simple and multiple linear equations for predicting θ_v using these soil parameters are presented in this paper. The volumetric water content of these 10 agricultural soils can be accurately predicted using these five soil properties, which are routinely measured in the field by professional field soil scientists.

Introduction

Soil is a porous media with pores of many sizes and shapes. Water that enters the soil either remains in the soil or percolates through it to lower depths in the soil profile. The size and shape of the pore space and the continuity of the pores determine the degree of water retention by the soil particles. Attempts have been made to formulate soil moisture constants to express differences in the water-holding capacity of soils. Baver (1956) included a historical perspective of the early concepts and hypotheses about soil-water relationships. Water retention was viewed as a tension of water films around particles, and early literature discussed the capillary tube concept, where water was thought to exist as a continuous and tightly stretched film around soil particles.

Briggs (1897) proposed that soil water could be classified as hygroscopic, capillary, and gravitational water. Buckingham (1907) introduced the idea of energy relationships in soil moisture retention. He envisioned that the flow of water through soil could be compared to the flow of heat through a metal bar, or the flow of electricity through a wire. The driving force was the difference in attraction for water between two portions of the soil not equally moist. He suggested the term "capillary potential," characterized by the Greek letter ψ, to express the attraction of the water for the soil at different soil moisture contents. The physics term "potential" was used because it defines the work that is necessary to bring a unit quantity of water from a given reference energy state to a different energy state. Gardner (1920) defined capillary potential as the work required to move a unit mass of water from a point where the potential is zero to the point in question. The liquid exerts a greater attraction to the soil surface than the air, causing a tension, which is a negative pressure.

Richards (1928) expanded the concepts of Gardner (1920) and measured the moisture content of various soils at different negative potentials. He showed that relative to fine-textured soils, coarse-textured soils exhibit a higher potential at low moisture contents, and fine-textured, compared to coarse-textured soils, contain much more water at the same potential. Richards and others developed porous plates and porous membrane techniques to determine the moisture-tension curves of soils. In recent years these measurement methodologies have been modified and improved, as described in Klute (1986).

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Soil-moisture constants are used to express soil-water relationships. The older literature discussed the hydroscopic coefficient, which represents the amount of water adsorbed on the surface of soil particles from an atmosphere of water vapor of known relative humidity. It is an index of the surface activity of the soil, but is of minimal interest. However, the field capacity and permanent wilting percentage (or wilting coefficient) are extensively used. The energy expression for water being held to soil surfaces has been the atmosphere or bars in the past; however, the SI unit of pressure, the Pascal (Pa), is now used. Table 1 shows the equivalents among expressions of soil-water potentials (ψ).

In the past, field capacity was often defined as the soil moisture content at –0.33 bars potential; however, we now define it as the amount of water remaining in a soil from 5–6 hours in very sandy soils to 1–2 days in loamy or finer textured soils, after being saturated and free drainage has ceased. The water potential at this point is generally about –10 kPa for sandy soils and –33 kPa for clayey soils. Usually the wilting point is defined as the water content in soils at about –1500 kPa potential. The difference in moisture content between field capacity and the wilting point is sometimes referred to as the available water-holding capacity of a soil. Hillel (1998) and Or and Wraith (1999) have explained soil water content and water potential relationships in great detail. Basic soils textbooks (Brady and Weil 1999; Miller and Gardiner 2001) also discuss this topic in various degrees of detail.

The objectives of this research were (1) to determine the volumetric water content of 10 irrigated Arizona soils ranging from sand to silty clay at –10, –33, –100, and –1500 kPa of potential; and (2) to correlate these moisture contents to the percent clay, percent sand, stickiness, plasticity, and bulk density of these 10 soils.

Table 1. The approximate equivalent units for expressing soil-water potential (ψ).

Height of unit column of water, cm	Soil water potential, bars	Soill water potential, kPa
0	0	0
10.2	–0.01	–1
102	–0.1	–10
306	–0.3	–30
1,020	–1.0	–100
15,300	–15	–1,500
21,700	–31	–3,100
102,000	–100	–10,000

Methodology

Undisturbed soil cores were collected for 10 irrigated Arizona agricultural soils using the soil core method described in Blake and Hartge (1986). Two methods were used to measure the volumetric water content on these cores: a Tempe cell with a hanging water column for low tensions < 250 cm of water, and a pressure chamber fitted with a porous ceramic plate for tensions > 250 cm of water (Klute 1986). The soils were saturated from below and then put under tension, and desorption curves were generated using the van Genuchten fitting program following Wraith and Or (1998).

The percent clay, percent sand, stickiness, and plasticity of each soil were determined independently by eight professional field soil scientists–soil surveyors employed by the Arizona Natural Resources Conservation Service and the University of Arizona, and the mean results for each parameter estimation were computed. The field method for determining soil texture and the percent clay and percent sand in a soil sample is explained in Thien (1979), Soil Survey Division Staff (1993), Brady and Weil (1999), and others.

The methodology for determining the stickiness and plasticity of soils is described in the USDA Soil Survey Manual (Soil Survey Division Staff 1993). Stickiness refers to the capacity of a soil to adhere to other objects, and plasticity is the degree to which puddled soil material is permanently deformed without rupturing by force applied continuously in any direction. There are four classes for each characteristic, and the description and terminology for placing them into one of the classes is presented in the Soil Survey Manual (Soil Survey Division Staff 1993: 178–179). In this study we asked soil scientists to make these estimates quantitatively. We assigned a numerical range to each class: 0 to 1, nonsticky or non-plastic; 1 to 2, slightly sticky or slightly plastic; 2 to 3, moderately sticky or moderately plastic, and 3 to 4, very sticky or very plastic. The soil scientists were asked to first place each soil into one of the four classes, and then record a number, indicating where it most likely fit in the 0–1, 1–2, 2–3, and 3–4 ranges. For example, if a soil was determined to be moderately plastic, and it is identified as being in the middle of that class, they recorded a 2.5. If the sample tended toward the slightly plastic class, they recorded a 2.2 or 2.3. There is no laboratory procedure for determining stickiness and plasticity. Bagour (2001) compared percent clay and percent sand laboratory analyses with the mean field estimations, and the R^2 coefficients of determination were 0.98 and 0.95 for percent clay and percent sand, respectively. However, the soil scientists overestimated the average percent sand content by 8.5 percent, whereas the average constant for percent clay was < 1 percent. Post et al. (1999) evaluated the skill of soil scientists to determine these soil properties, and this should be known when using field estimations.

The absolute bulk density was measured using the core method (Blake and Hartge 1986); however we also assigned a bulk density "rating" to each soil. Three generalized figures in the Soil Survey Manual (Soil Survey Division Staff 1993: 110) show the relationships between soil texture and the measured bulk density. The bulk density ratings are low, medium, and high. We assigned a code of 0–1 to low, 1–2 to medium, and 2–3 to high bulk densities, using these three figures. This involved the interpolation of the iso-bulk density lines noted on the figures. The rationale for using this rating was to identify relative compactness or density, rather than using absolute bulk density measurements. Using this relative scale might make it easier for field soil scientists to quantify soil bulk density.

The percentage of volumetric water content (θ_v) at a given soil-water potential (ψ) was compared to the five soil properties using the following statistical parameters. The Pearson correlation coefficient, r, measures the strength of a linear relationship between two variables using the following formula:

$$r = \sum_{i=1}^{N} \frac{(x_1 - x)(y_1 - y)}{(N-1)S_x S_y}$$

where N is the sample size and S_x and S_y are the standard deviations of the two variables (SPSS Base 10, 1999).

The curve estimation procedure was used to produce the best curve regression statistics, and plots for 11 curve estimation regression models were evaluated. A separate model was produced for each dependent variable. When there is only one independent variable, r is the simple correlation between the dependent and independent variable. The coefficient of determination, R^2, can also be computed as follows:

$$R^2 = \frac{SSR}{SST}$$

where SSR is the regression sum of squares measuring the variability in the response variable attributed to the model, and SST is the sum of squares corrected for the mean of the response variable (which measures the total variability in the response variable). For multiple regression models, R is the correlation between the observed and predicted values of the dependent variable. For this study, the correlation between θ_v at a defined soil moisture tension and the five soil variables was evaluated. Because the sample estimate of R^2 tends to be an overestimate of the population parameter, the adjusted R^2 is used to compensate for this optimistic bias. It is a function of R^2 adjusted $\left(R^2_{adj}\right)$ by the number of variables in the model and sample size:

$$R^2_{adj} = 1 - \frac{residual\ sum\ of\ squares / (N-P-1)}{total\ sum\ of\ squares / (N-1)}$$

The value of R^2_{adj} is always smaller than the corresponding R^2 (SPSS Base 10, 1999).

Another statistic used to aid in the selection of a final model is called the Mallow's C_p, defined as

$$C_p = \left(\frac{SSE_p}{MSE}\right) + 2p - n + 1$$

where MSE is the mean square error for full model, and SSE_p is the sum of squares error for a model with p parameters (not including the intercept; Freund and Littell 1991). This model chooses the maximum R^2_{adj}, which gives the smallest C_p and most closely approximates the number of parameters in the model.

Multiple regression analysis was used to predict the volumetric water content from the five soil variables at the various soil moisture potentials. The multiple regression method begins by entering all of the variables into the model. The output analysis of the regressions procedure was generated using SPSS Base, 1999 software, Version 10.

Results and Discussion

Table 2 presents the mean field estimations for percent sand, percent clay, stickiness, and plasticity as measured by the eight professional soil scientists for the 10 soils we studied. The absolute bulk density measured in the laboratory and the bulk density "rating" are also listed. Table 3 lists the textural class and the volumetric water content for each soil at –10, –33, –100, and –1500 kPa of soil

Table 2. Mean of field estimations by eight professional soil scientists for selected soil morphologic properties and bulk density (BD), and total pore space (TPS) data.

Soil series and Textural class	% Sand	% Clay	Plasticity	Stickiness	Absolute BD*	BD Rating	TPS %
Brazito (S)	98.0	1.0	0.05	0.05	1.45	0.9	45.3
Superstition (LS)	83.6	4.4	0.60	0.28	1.76	2.1	33.6
Vinton (FSL)	74.9	7.1	0.64	0.74	1.53	1.5	42.3
Casa Grande (SL)	67.1	19.8	2.04	2.00	1.60	2.0	39.6
Gila (VFSL)	62.7	11.9	1.36	1.20	1.48	1.5	44.1
Casa Grande (SCL)	61.4	27.8	3.09	2.84	1.53	1.8	42.3
Anthony (VFSL)	58.0	16.3	1.59	1.35	1.56	1.7	41.1
Grabe (L)	28.3	19.9	2.44	1.89	1.49	2.2	43.8
Pima (SiCl)	6.5	38.0	3.61	3.09	1.79	3.0	**
Gadsen (SiC)	8.7	53.6	3.61	3.06	1.47	2.5	44.5

*Expressed in g/cm^3.
**Not computed, as explained in text.

Table 3. Volumetric water content (θ_v) measured on undisturbed soil cores at –10, –33, –100, and –1500 kPa of soil-water potential (ψ).

Soil series and Textural class	Volumetric water content at –10 kPa	–33 kPa	–100 kPa	–1500 kPa
Brazito (S)	0.1356*	0.0661	0.0503	0.0448
Superstition (LS)	0.1500	0.1080	0.0885	0.0720
Vinton (FSL)	0.2083	0.1637	0.1386	0.1098
Casa Grande (SL)	0.2338	0.1860	0.1600	0.1311
Gila (VFSL)	0.2492	0.2064	0.1803	0.1458
Casa Grande (SCL)	0.2764	0.2292	0.2033	0.1744
Anthony (VFSL)	0.2644	0.2291	0.2046	0.1671
Grabe (L)	0.3155	0.2750	0.2513	0.2230
Pima (SiCl)	0.3522	0.3275	0.3082	0.2814
Gadsen (SiC)	0.4411	0.4219	0.4006	0.3650

*Decimal fraction of θ_v per unit depth of soil, such as cm/cm or in/in.

moisture potential. The total pore space is also listed, which was calculated as follows:

Total pore space =

$$100 - \left(\frac{\text{bulk density, g /cm}^3}{\text{particle density, g /cm}^3} \text{x } 100 \right).$$

We assumed that the particle density was 2.65 g/cm^3. All soils except Brazito sand had moderate to high bulk densities, and the Pima soil was very dense. The structure was mostly massive or weak, medium blocky, except for the Brazito which was single grained. The Pima soil had an initial bulk density of 1.79 g/cm^3 when sampled, which was very dense for a silty clay loam texture. It swelled significantly when wetted in the laboratory; thus the porosity of this soil changed. For this reason, we did not list a total pore space value for this soil in Table 2.

Soil water content is commonly measured on a weight or mass basis (θ_m), rather than the volume basis reported here. θ_m is calculated as follows:

$$\theta_m = \frac{\textit{mass of moist soil} - \textit{mass oven dry soil}}{\textit{mass oven dry soil}}$$

This is multiplied by 100 if θ_m is expressed as a percentage.

The volumetric water content (θ_v) can be computed as follows:

$$\theta_v = \theta_m \left(\frac{\textit{bulk density of soil, } g/cm^3}{\textit{density of water, } g/cm^3} \right)$$

The density of water is 1.0 g/cm^3, which cancels these units, so $\theta_v = (\theta_m)$ (bulk density – unitless). A field person has to know the bulk density of the soil, or be able to convert the bulk density rating system into the appropriate g/cm^3, to convert θ_m to θ_v. The following ranges of bulk density from the Soil Survey Manual (Soil Survey Division Staff 1993: 109) can be used as a general guide for Arizona soils with no rock fragments. All textural classes except the sandy textures with > 1.5 g/cm^3 are rated high; sandy textures > 1.7 are considered high. Bulk densities of 1.3 to 1.5 g/cm^3 are usually medium, and from 1.1 to 1.3 g/cm^3 are low ratings.

Table 4 presents the relationships between soil properties and the volumetric water content at –10, –33, –100, and –1500 kPa of pressure. The correlation coefficients for percent clay for the four water potentials average $r = 0.95$, and for sand, $r = -0.96$. The mean r values for stickiness and plasticity were 0.87 and 0.90, respectively. There was

Table 4. Pearson correlations (r), R^2, and simple linear regression relationships between soil properties and for predicting volumetric water content at –10, –33, –100, and –1500 kPa of soil-water potentials.

Soil properties	r	R^2	Simple linear regression
$\psi = -10$ kPa			
Clay	0.949	0.901	$\theta_v = 0.0054x + 0.1554$
Sand	–0.951	0.904	$\theta_v = -0.0028x + 0.4181$
Stickiness	0.882	0.777	$\theta_v = 0.0240x + 0.1432$
Plasticity	0.908	0.824	$\theta_v = 0.0652x + 0.1386$
Bulk density – absolute		0.005	No correlation
Bulk density rating	0.720	0.522	$\theta_v = 0.1133x + 0.0451$
$\psi = -33$ kPa			
Clay	0.948	0.899	$\theta_v = 0.0060x + 0.1007$
Sand	–0.958	0.915	$\theta_v = -0.0032x + 0.3978$
Stickiness	0.874	0.765	$\theta_v = 0.0810x + 0.087$
Plasticity	0.904	0.816	$\theta_v = 0.0730x + 0.0823$
Bulk density – absolute		0.000	No correlation
Bulk density rating	0.753	0.568	$\theta_v = 0.1329x - 0.0352$
$\psi = -100$ kPa			
Clay	0.949	0.900	$\theta_v = 0.0060x + 0.0783$
Sand	–0.960	0.922	$\theta_v = -0.0032x + 0.3747$
Stickiness	0.868	0.753	$\theta_v = 0.0799x + 0.0667$
Plasticity	0.899	0.809	$\theta_v = 0.0725x + 0.0607$
Bulk density – absolute		0.000	No correlation
Bulk density rating	0.757	0.573	$\theta_v = 0.1332x - 0.0572$
$\psi = -1500$ kPa			
Clay	0.956	0.913	$\theta_v = 0.00570x + 0.0575$
Sand	–0.965	0.931	$\theta_v = -0.0030x + 0.3380$
Stickiness	0.865	0.748	$\theta_v = 0.0750x + 0.0477$
Plasticity	0.901	0.811	$\theta_v = 0.0683x + 0.0415$
Bulk density – absolute		0.000	No correlation
Bulk density rating	0.766	0.589	$\theta_v = 0.1268x - 0.0725$

no relationship between the absolute bulk density and θ_v, which is expected because different textured soils have similar bulk densities. However, the r value for bulk density ratings and θ_v was 0.75. Clearly, percent sand and percent clay are most strongly correlated to θ_v at different soil moisture potentials, with clay being positive and sand a negative relationship. Figures 1 and 2 show two scattergrams of these relationships. Table 4 lists the simple linear regression equations that relate the soil properties to the volumetric water content at –10, –33, –100, and –1500 kPa of soil-water potential.

Although the simple linear correlations for percent clay and percent sand are very significant, we computed multiple linear regression equations to predict the volumetric water. Table 5 lists the multiple regression equations, and there are two equations for each of the four soil-water potentials, listed as equations 1 and 2. All equations include percent clay, percent sand, stickiness, and plasticity. Equation 1 uses the bulk density rating, and equation 2 uses the measured bulk density. The R^2 for these eight equations ranged from 0.981 to 0.996 and the R^2 adj. ranged from 0.947 to 0.990. The Mallow's C_p was 6.00 for all equations. Obviously, θ_v can be predicted very accurately using these equations.

Conclusions

This research has shown that four soil properties, percent clay, percent sand, stickiness, and plasticity, routinely determined in the field by professional soil scientists, can be used to accurately predict the volumetric water content at different soil moisture potentials. The field skill of the person determining these soil properties would obviously affect the results. Percent clay and per-

Table 5. Multiple linear regression equations for predicting the volumetric water content at –10, –33, –100, and –1500 kPa of soil–water potentials.

	R^2	R^2_{adj}
$\psi = -10$ kPa		
1 $\theta_v = + 0.002884$ (% clay) $- 0.00236$ (% sand) $+ 0.001979$ (stickiness) $- 0.00528$ (plasticity) $- 0.0458$ (bulk density rating) $+ 0.430$	0.982	0.959
2 $\theta_v = + 0.002358$ (% clay) $- 0.001776$ (% sand) $+ 0.008770$ (stickiness) $- 0.006234$ (plasticity) $- 0.141$ (bulk density) $+ 0.531$	0.990	0.976
$\psi = -33$ kPa		
1 $\theta_v = + 0.0034061$ (% clay) $- 0.00256$ (% sand) $+ 0.006574$ (stickiness) $- 0.0164$ (plasticity) $- 0.00324$ (bulk density rating) $+ 0.375$	0.976	0.947
2 $\theta_v = + 0.002984$ (% clay) $- 0.002135$ (% sand) $+ 0.009751$ (stickiness) $- 0.01477$ (plasticity) $- 0.109$ (bulk density) $+ 0.460$	0.981	0.958
$\psi = -100$ kPa		
1 $\theta_v = + 0.003641$ (% clay) $- 0.00252$ (% sand) $+ 0.00571$ (stickiness) $- 0.00872$ (plasticity) $- 0.0319$ (bulk density rating) $+ 0.352$	0.981	0.957
2 $\theta_v = + 0.003255$ (% clay) $- 0.002113$ (% sand) $+ 0.001651$ (stickiness) $- 0.008461$ (plasticity) $- 0.102$ (bulk density) $+ 0.428$	0.985	0.966
$\psi = -1500$ kPa		
1 $\theta_v = + 0.00378$ (% clay) $- 0.00222$ (% sand) $+ 0.0284$ (stickiness) $- 0.01062$ (plasticity) $- 0.0293$ (bulk density rating) $+ 0.301$	0.994	0.987
2 $\theta_v = + 0.003476$ (% clay) $- 0.001851$ (% sand) $+ 0.02306$ (stickiness) $+ 0.008553$ (plasticity) $- 0.08412$ (bulk density) $+ 0.357$	0.996	0.990

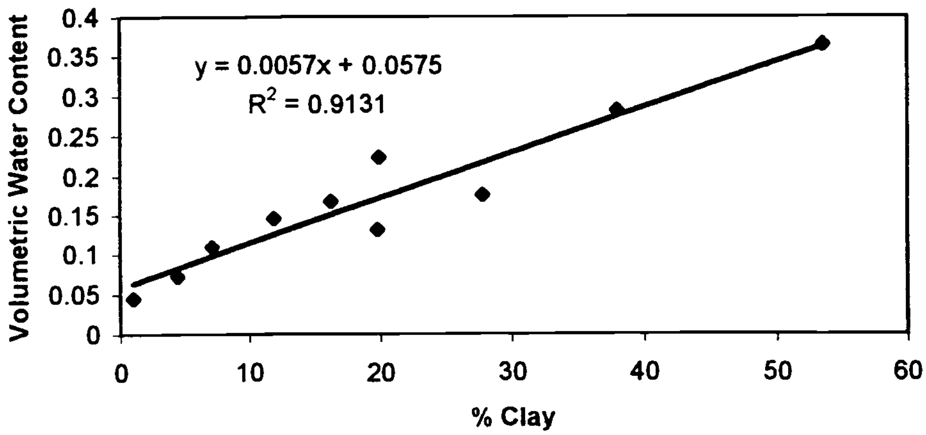

Figure 1. Simple linear regression between volumetric water content at –1500 kPa soil water potential and percent clay.

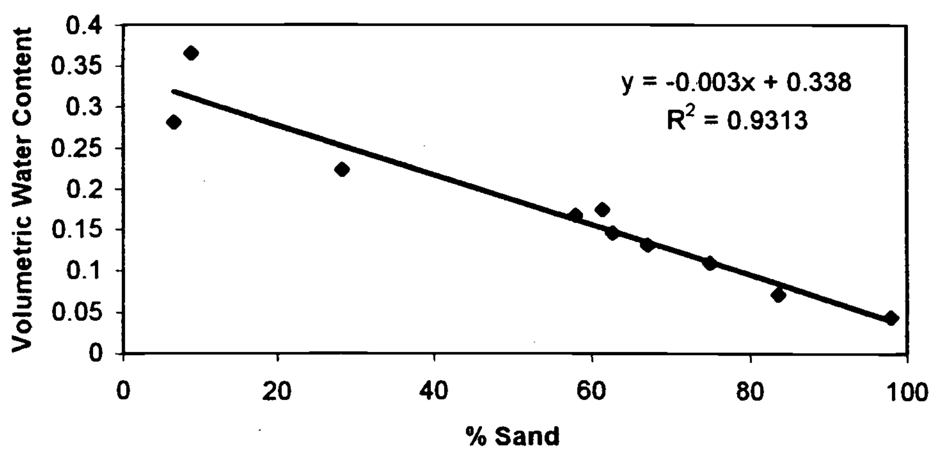

Figure 2. Simple linear regression between volumetric water content at –1500 kPa soil water potential and percent sand.

cent sand are the two most important properties; however, stickiness and plasticity were also useful. The bulk density or compactness of the soil is also important, and an accurate estimate of this property should also be included. The regression equations and correlations presented in this paper were for Arizona irrigated soils that contained no rock fragments. Further studies are needed to evaluate relationships such as these for other soils, particularly rangeland or forest soils that have many rock fragments present.

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ANALYZING THE FEASIBILITY FOR REROUTING THE RIO DE FLAG IN FLAGSTAFF, ARIZONA

Boris Poff, Richard Tappan, and Loretta G. Morgan*

Abstract

A significant portion of the city of Flagstaff, Arizona, has been built on the Rio de Flag floodplain. A 100-year flood would result in economic, social, environmental, and regional impacts and damages that can devastate the community. Despite more than 20 in-depth studies about the flooding potential of the Rio de Flag, the city has taken very few precautions, except those required by the Federal Emergency Management Agency's (FEMA) flood insurance policy, issued in 1983. The draft feasibility report and environmental impact statement recommends the elimination of an existing detention pond and the construction of both open and closed channels as well as a new larger detention pond. Shortcomings of this approach include a paucity of data about the streamflow history of the Rio de Flag, and inadequate consideration of greenbelt issues.

Study Area

Flagstaff is situated on the Colorado Plateau just south of the San Francisco Mountains at an elevation of 6,900 feet (Fenneman 1931). Despite its location in a semi-arid climate, the altitude allows Flagstaff to experience four seasons that include hot summers and cold winters. The average annual precipitation is 20 inches, with nearly half occurring as snowfall (Hill 1988). Monsoonal winds bring moisture from the Gulf of California during the late summer and early autumn, causing heavy precipitation events (Fenneman 1931).

Most of downtown Flagstaff and the area along the Rio de Flag lie within a 100-year floodplain. Flooding in this area would affect many residences, businesses, and schools. A major flood in the downtown area occurred in 1923 and again in March of 1982 (Sellers 1985; Compass 2000). The latter inundated Flagstaff's streets, residential areas, and mobile home parks to a depth of several feet (Sellers 1985). The east side of Flagstaff was severely flooded in the 1950s by a flash flood, requiring a complete renovation of the storm drainage system along Route 66 and the Fourth Street commercial neighborhoods (Cline 1990). The last major flood in terms of discharge rate occurred in 1938, and on a volume basis in 1993 (Compass 2000). The past history of flooding within the city indicates that flooding events can occur during any season of the year.

Winter storms generally cover large areas and are usually of long duration. Summer storms are usually associated with tropical depressions (dissipating tropical cyclones) that have a short duration and cover a large area. Local storms, which are usually thunderstorms, generally occur during the summer, with high intensity and short duration. Intense short-duration rainfalls, heavy snowpack with ripe melting conditions, severe rainfall on melting snow, warm rain on snow during the winter with frozen ground conditions, or a series of storms can all lead to substantial runoff and flooding (Compass 2000).

Annual temperature extremes in the Flagstaff area typically range from 33°C to –18°C. The yearly average high and low temperatures are 16°C and 1°C, respectively. The prevailing winds are from the southwest with an average speed of 12–15 kph (Hill et al. 1988).

The geology around Flagstaff consists of a mixture of volcanic and sedimentary rocks. North of Santa Fe Avenue, the city is built on mostly volcanic rock, whereas the south side of the town lies mostly on sedimentary rock from the Kaibab and Moenkopi Formations. The nature of the exposed rocks directly affects the degree of runoff or infiltration. The rock types exposed at the surface in the Rio de Flag channel contribute to rapid infiltration of surface water through either their porosity (volcanic cinders and basalt) or their fractures (lava and calcareous sediment rock). The Soil Conservation Service (SCS) has established

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four hydrological soil groups according to their runoff-producing characteristics (Hill et al. 1988). Group A soils exhibit little to no runoff, and Group D soils exhibit abundant runoff. The Rio de Flag soils are in SCS group D. However, the watershed soils are mostly high in clay content (groups B and C), and when they freeze in the winter their infiltration rate is virtually zero (Hill 1988).

Streamflow Conditions

The Rio de Flag has not been gauged on a continuous basis, making long-term or current flow conditions unavailable. In 1969 the USGS established a network of gauging stations (Hill et al. 1988); the three gauges of interest here are located upstream of Flagstaff at Hidden Hollow Road, downtown, and downstream at Interstate 40. Figure 1 shows the annual peak discharges for these gauges (Hill et al. 1988).

USGS analysis indicates that discharges have recurrence intervals of 2, 5, 10, and 25 years. Peak flow conditions originate mainly in the urban areas, where there are few storm sewers, channel improvements, or storage areas, resulting in variations in computed discharge (Hill et al. 1988).

Problem Statement

According to the "Rio de Flag Draft Feasibility Report and Environmental Impact Statement" (Compass 2000), the major problem in the area is flooding, which results in inundation damage, railroad damage, emergency response costs, and transportation delays. Nearly half of the 100-year floodplain along the Rio de Flag is zoned as residential, with commercial accounting for nearly a quarter. Development within the floodplain is extensive (Compass 2000). The downtown and south side areas contain numerous registered historic structures, some over a hundred years old (Compass 2000; Cline 1990). If no action is taken the city of Flagstaff will continue to be subject to significant economic, social, and environmental consequences from severe floods. Approximately 1,500 existing structures, worth about $385 million, could suffer about $93 million worth of damage from a 1 percent flood event. A significant portion of Northern Arizona University lies within the floodplain, and during severe flood events the university would incur closing and other disruptions and physical damage to facilities and historic buildings on campus. Numerous residential, commercial, downtown business, tourism, and industrial properties would remain at risk.

Planning Objectives and Alternatives

The draft feasibility report and environmental impact study provides four important planning objectives: (1) Minimize flood damages to residential, commercial, public, industrial, and historic property. (2) Develop a comprehensive plan. (3) Provide consistency with local initiatives and the cultural and environmental character of the community, including aesthetics. (4) Protect and improve environmental and cultural resources. To achieve these objectives, seven alternatives were presented to the city.

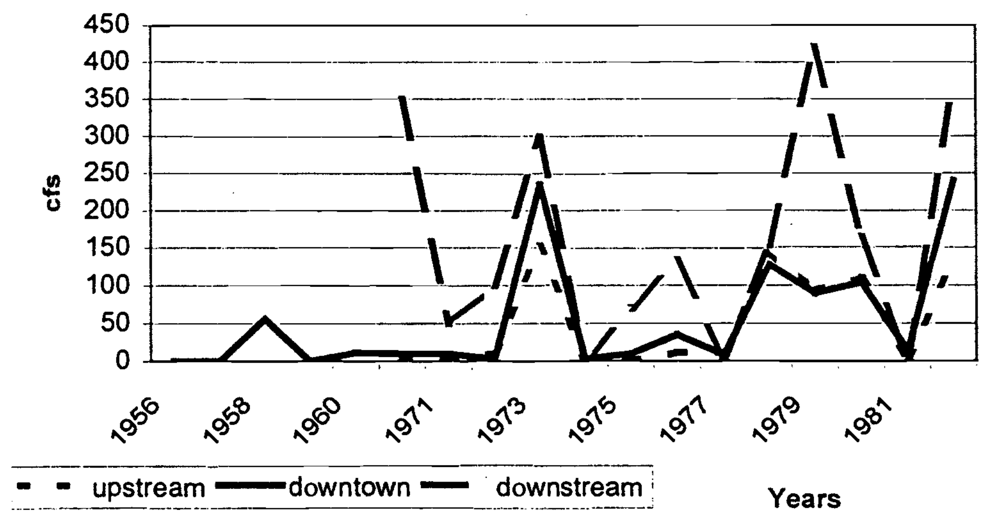

Figure 1. Annual peak discharges for three gauges used by the USGS around Flagstaff, Arizona.

Alternative 1 calls for the construction of flood walls at the Thorpe Park detention basin to minimize outflow into the Rio de Flag. The construction of a detention basin on Clay Avenue Wash west of Flagstaff is also included as a part of this alternative.

Alternative 2 consists of the construction of the flood walls at Thorpe Park as well as extensive channel improvement. The latter would include stretches of open and closed concrete, with earthen bottoms and natural rock revetment. Portions of the current Rio de Flag alignment would remain, with part being realigned to parallel the railroad tracks until the historic Rio de Flag is reached just downstream of Butler Avenue.

Alternative 3 proposes construction of the detention basin on Clay Avenue Wash and a rectangular concrete channel upstream of the Railroad Springs development. Channel improvements of the Rio de Flag from this point are similar to those presented in Alternative 2.

Alternative 4 represents a full channelization plan, with no detention basins. The improvement of Clay Avenue Wash, as presented in Alternative 3, is also included as a part of this option.

Alternative 5 includes both the construction of the detention basins and channel improvements along Clay Avenue Wash and the Rio de Flag upstream of Birch Street. Further channel improvement between the confluence of the two washes and Butler Avenue is also planned.

Alternatives 6A, 6B, and 7 were presented after reevaluations of the engineering, design, and cost of Alternatives 1–5. Alternatives 6A and 6B are similar except for channel treatments along the Rio de Flag between Cherry and Birch Avenues. In Alternative 6A this part of the channel is riprap construction, whereas Alternative 6B calls for a covered concrete arch channel. Both exclude construction of the Thorpe Park detention basin.

Alternative 7 consists of constructing one detention basin upstream of Thorpe Park in addition to the two previously presented detention basins and channel improvements along the Rio de Flag.

Selected Plan

Alternative 6B was selected by the city of Flagstaff because it would "provide flood protection along the Rio de Flag's Downtown Reach and would also reduce flooding along the Clay Avenue Wash" (Compass 2000). Except for portions of Clay Avenue Wash just downstream of the new detention basin, its construction in combination with the channel modifications along Clay Avenue Wash and the Rio de Flag would prevent residual flooding during a 100-year event.

The detention basin along Clay Avenue Wash would allow for the discharge to be over a period of 50 to 60 hours after the basin reaches maximum storage—allowing for an extended period of flow in downstream channels. The design includes an emergency spillway for flood events in excess of the 100-year level of protection and also to prevent basin failure.

Changes at Thorpe Park will consist of new structures on the southern and eastern boundaries of the park. The structures along the southern boundary will be small embankment-wingwalls to direct flows into the existing channel downstream and away from existing development. The eastern boundary of the park will have a series of flood walls to ensure that water does not overtop this part of Thorpe Park, which would flood the adjacent residential area to the east and south. Both the southern and eastern structures will be constructed with aesthetically pleasing local rock fascia, with appropriate native trees and other vegetation (Compass 2000).

Channel modification will be extensive, including stretches of open and closed concrete with earthen bottoms and natural rock revetment along both Clay Avenue Wash and the Rio de Flag (Figures 2 and 3). This alternative also includes adding a greenbelt immediately after the covered concrete streambed (Figure 4). The greenbelt would be located in an established commercial zone to help distribute flood flows before entering the canyon southeast of historic downtown.

Discussion

There are several problems with the placement of concrete structures in the streambed. The placement of open concrete channels, riprap channels, and concrete culverts upstream of a significant greenbelt could be hazardous to the longevity of the downstream greenbelt habitat. Concerns that should be addressed before implementation of this project include reduction of the local water table and water quality due to channelization structures impeding infiltration and capturing non-source point pollution.

The two detention basins at Thorpe Park and Clay Avenue are expected to help reduce the peak flow rate by redistributing runoff over time and enhancing groundwater during and shortly after large storm events (Hall 1984). These basins will

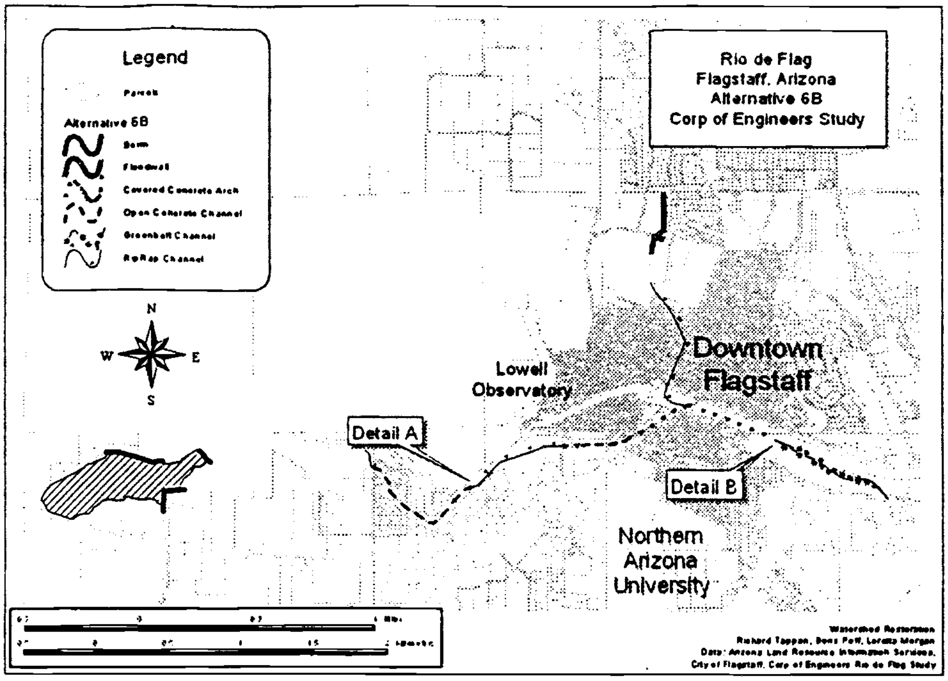

Figure 2. The proposed Alternative B.

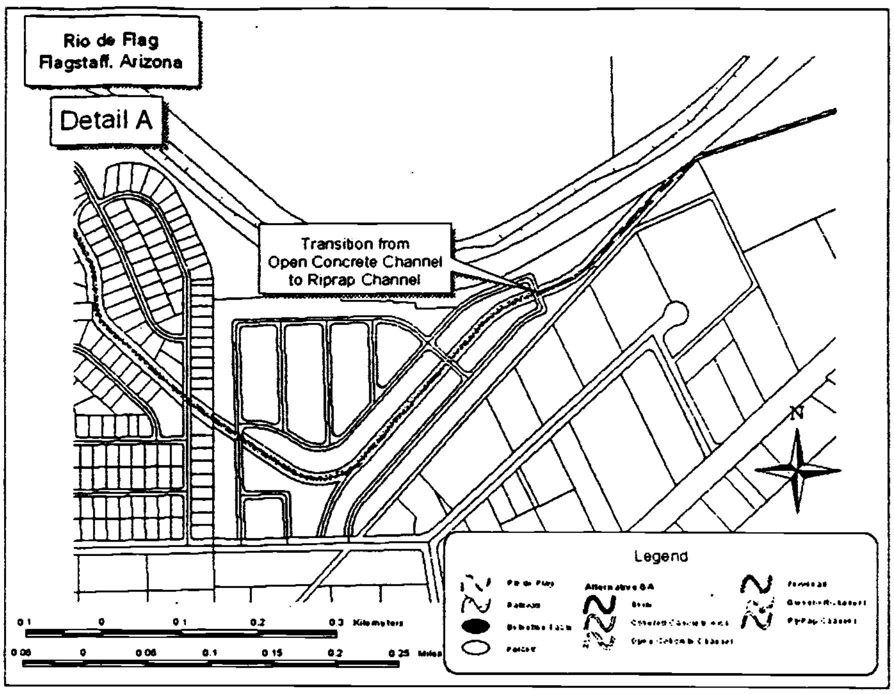

Figure 3. Detail of the proposed Alternative B.

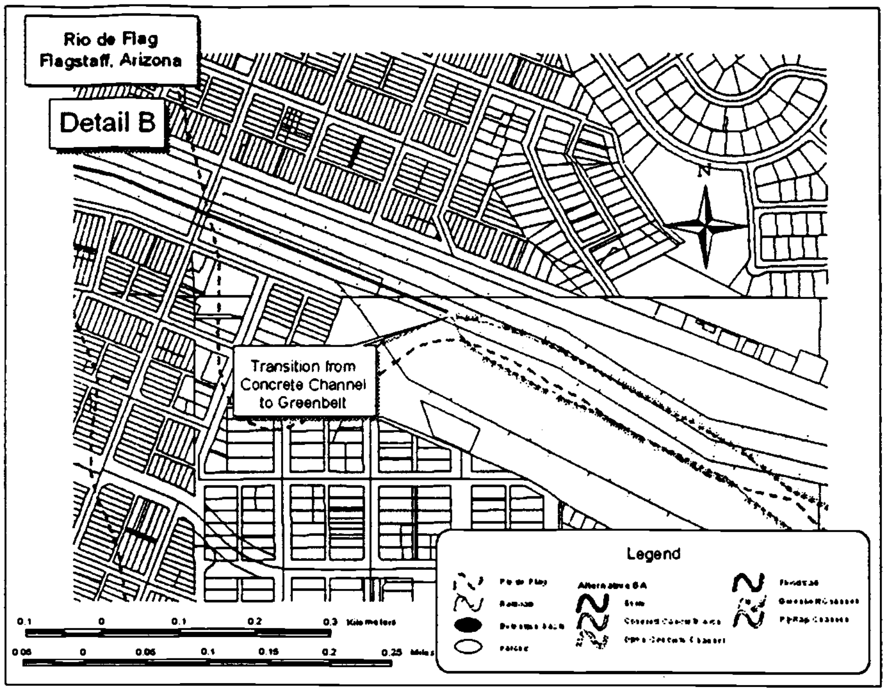

Figure 4. Detail of the proposed Alternative B.

also help waterborne solids to settle out of solution and thereby improve water quality. However, most of this benefit could be negated by the multitude of small to medium scale storms that will carry the bulk of the pollutant load downstream (Gordon et al. 1993).

The basins are located directly upstream from the channelized sections of the Rio de Flag that are of greatest concern to the well-being of the downstream greenbelt. Clay Avenue Wash, the major tributary, combines with the Rio de Flag south of Thorpe Park, posing a considerable problem in the center of town at San Francisco Street. At maximum carrying capacity the two washes join together in a concrete culvert that can accommodate only two-thirds of the total water. Clay Avenue Wash consists of riprap and open concrete channels with a cross section of 132 square feet. Open concrete structures and open channel riprap are proposed for the wash south of Thorpe Park with a cross section of 192 square feet. Both of these washes lead into a culvert 204 square feet in area. It is doubtful that the culvert is large enough to handle a 100-year flood event.

The concrete culvert at San Francisco Street is the last concrete emplacement before the greenbelt. Flow velocities entering this fragile area are most likely greater than the greenbelt's capacity to handle without creating serious erosion. Channelization, or straightening and lining of a stream corridor, increases channel slope and thus water velocity and sediment transport capacity (Gordon et al. 1993). If the stream is starved for sediment due to the upstream catchment basins, erosion problems may result downstream (Riley 1998). Adjusting the sediment budget for the stream could cause erosion in the downstream reaches, slowly working back upstream and causing bank collapse and bank erosion, and locally enlarging the channel width by two or three times (Riley 1998).

According to geomorphologist Robert Curry, streams that overtop their channels during high stage flow "with their ... higher velocities will ... often assume a braided or meandering pattern much to the detriment of the city established along its banks" (Riley 1998). The separation of a channelized stream from the floodplain will hinder the ability of the floodplain to store, release, and direct waters, thereby increasing the downstream flood peak (Gordon et al. 1993).

The greenbelt downstream of the structural channel modifications will help to slow the stream flow moving toward the east side of town, as well as acting as a recharge area for the local water table. There is potential that the created natural beauty will serve as a recreational area given that facilities will be provided. However, the proposed greenbelt is too short a corridor for the flood volume that will be transported through the area. As the flood waters enter the greenbelt, the velocities coming out of the open concrete channel and concrete culvert have the potential to strip the entrance of the greenbelt of all vegetation and redeposit the material further down where velocities will be decreasing (Hall 1984). This in turn might have the effect of damming the stream channel and driving the flood waters out of the greenbelt and into the neighboring areas.

Additional Suggestions

Regarding Alternative 6B, it is unknown how much of a possible flood would be contained within the proposed detention ponds and concrete streambeds, as discussed earlier, because no exact flow rate data are available for the Rio de Flag. It is therefore difficult to make any useful and reasonable suggestions other than those proposed in the alternative projects listed above.

Also, the planning of the proposed greenbelt appears to be inadequate. The greenbelt will be located in an industrial area, but no studies have been conducted on the possibility of chemical pollutants, which are likely to occur in an area after 100 years of industrial use. The greenbelt is designed to slow the floodwaters and recharge the groundwater, but it may inadvertently pollute the watershed downstream, as well as the groundwater. Further, it is doubtful that the Rio de Flag will provide enough moisture to sustain a greenbelt adequate to minimize the erosion concerns mentioned in the discussion. A combination of native sedge grasses (*Poa* spp.) and willows (*Salix* spp.) would be ideal for slowing down the potential velocity of the floodwaters (Kolb and Moore 1999; Briggs 1996). However, these species require a certain amount of moisture, which will not be guaranteed. Would the city of Flagstaff be willing to artificially irrigate the greenbelt if it becomes necessary? These issues should be addressed before any alternative is implemented.

Further, the feasibility report lists several planning objectives, including "the protection and improvement of environmental and cultural resources as well as the provision of consistency with local initiatives and the cultural and environmental character of the community including aesthetics," but there is no indication that these planning objectives were considered. Instead it appears that the only planning objective addressed was how to protect most of the city while spending the least amount of money. Perhaps a survey could assess the views of the local residents about the aesthetics of a concrete channel compared to the other alternatives. In other parts of the country communities are restoring natural channels where the U.S Army Corps of Engineers has constructed concrete channels, often at greater expense than the original project.

One possible alternative would be building the detention basins at Clay Avenue Wash and above Thorpe Park as well as improving the existing detention basin. Instead of a concrete channel, perhaps a clean-up and restoration of the existing channels of the Rio de Flag should be considered. Such an alternative addresses all of the planning objectives outlined in the feasibility study (Riley 1998). Geographers, hydrologists, and hydraulic experts have found that such traditional engineering techniques have unanticipated performance problems (Riley 1998), as mentioned above.

Conclusion

The U.S. Army Corps of Engineers provided several alternatives to help the city of Flagstaff to curb the risk of future potential flooding. Alternative 6B, which was selected by the city, appears to be the most feasible of the alternatives provided. It suggests two detention ponds upstream of the floodplain as well as both open and closed concrete channelization of the stream, followed by a greenbelt. However, there are still several shortcomings. Some of these shortcomings are caused by the lack of information on the volume and intensity of flow that can occur in the Rio de Flag and its tributaries. This missing data has led the U.S. Army Corps of Engineers to rely on their best estimates of future flows. The other shortcoming is the inadequate planning for a greenbelt, which might lead to pollution and further flooding downstream of the modified channel. Further studies should be conducted on the effects that the selected alternative will have downstream and which type of greenbelt vegetation should be implemented and how. None of the alternatives provided, including Alternative 6B, address all of the planning objectives outlined by the feasibility study. An alternative that includes a total of three detention ponds, but refrains from the construction of any concrete channels, might be a better solution.

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RELATIONSHIP OF FINE SEDIMENT AND TWO NATIVE SOUTHWESTERN FISH SPECIES

John N. Rinne*

In the southwestern United States, the native fish fauna is low in diversity and is composed primarily (95%) of cypriniform (minnow and sucker) species (Minckley 1973; Rinne and Minckley 1991). All native species have declined in range and numbers in the past half century (Miller 1961; Rinne 1994, 1996). As a result, most of the native fauna is either federally or state listed (Williams et al. 1989; Minckley and Deacon 1991; Rinne and Minckley 1991). Spikedace (*Meda fulgida*) and loach minnow (*Rhinichthys* [*Tiaroga*] *cobitis*) are two such natives. These two federally threatened species are restricted to the Gila River basin in Arizona and New Mexico, and have declined dramatically in range and numbers (Minckley 1973; Rinne and Minckley 1991; U.S. Fish and Wildlife Service 1990a, 1990b).

Fine sediment (< 2 mm) is a natural component of bed load in streams (Hynes 1972:23–24). Fines can affect aquatic environments and their inhabitants in both suspended and depositional form. Considerable literature has addressed the effects of fines on salmonids and their habitats (e.g. Meehan 1991), but most information is on the depositional, substrate, or bedload state of fines. By contrast, there is a complete lack of information on the effects of the fine component of stream substrates on non-salmonid species.

Anthropogenic activities across the southwestern landscape are thought to indirectly impact native fishes and their habitats (Rinne et al. 1998, 2001). In the Southwest, livestock grazing has generally been implicated as the major, extrinsic contributor to native fish decline in general, and specifically to the decline of these two threatened species. However, there is no information on the mechanisms of the impact from grazing on native fishes (Rinne 1999a, 1999b, 2000; Medina and Rinne 1999).

The fine sediment produced by livestock grazing and its effect on spawning habitat (substrate) is frequently offered as the primary mechanism of impact on these two species. However, a paradox exists. Spikedace have declined and have remained absent in one southwestern desert river for a few years despite livestock grazing removal (Rinne 1999a; Medina and Rinne 1999). By contrast, both spikedace and loach minnow are sustaining themselves in another reach of desert river where livestock grazing is present (Rinne et al. 2001). Biological consultation is ongoing for many grazing allotments with streams and rivers in U.S. Forest Service Region 3 that have historically sustained or currently contain the species.

Because of (1) the status of these two native fish species and (2) the suggestion that grazing indirectly and negatively impacts both species through fine sediment production, it is timely to examine trends in distribution and abundance data for both, relative to sediment fine composition of stream substrates. Conceptually, such an approach should commence to define the relationships and the probable impact of fine sediment on spikedace and loach minnow.

The major objectives of this paper are to examine and define (1) the substrate composition of spikedace and loach minnow habitat, (2) the fine component of these substrates where spikedace and loach minnow are present and absent, and (3) the probability that sediment fines have been a primary causative factor in the decline of these two species.

Methods and Study Areas

Spikedace were collected with both seines and backpack DC electrofishing techniques depending upon habitats sampled (Stefferud and Rinne 1995; Rinne et al. 2001). Loach minnow were collected primarily by electrofishing into block seines positioned 2–4 m downstream of initial points of sampling. Individuals were immobilized and washed downstream into the net. All fish were removed, enumerated, measured, and returned alive to the stream.

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Substrates were characterized by pebble count methodology (Bevenger and King 1995). Water velocity was measured with a direct readout flow meter and depths with a meter rule. The study areas included the upper Verde River (Stefferud and Rinne 1995) in west-central Arizona and the upper Gila River (Rinne et al. 2001) in southwestern New Mexico.

Fines and Fishes

Substrates Utilized

Substrates are a reflection, in part, of gradients and velocities of aquatic habitats (Hynes 1972: 11). Spikedace were found over sand (< 2 mm, 21%) to gravel (3–64 mm, 55%) substrates in the glide-run and low-gradient riffle habitats in both the upper Verde (Rinne and Stefferud 1996) and the upper Gila (Rinne and Deason 2000). Loach minnow occupied only high-gradient riffle habitat composed of gravel (38%) and pebble (65–127 mm; 32%) substrate. Rinne (1991) reported on substrates occupied by spikedace in several southwestern rivers and streams, however numeric indexes of 1 to 5 (sand-gravel-pebble-cobble-boulder) were utilized to define substrate habitats occupied. Spikedace was found over quite variable substrates depending on stream size. Rinne (1989) reported loach minnow also occupying a wide range of substrates from gravel to cobble in the same suite of streams in Arizona and New Mexico.

Fine Component of Occupied Habitats

Rinne and Stefferud (1996) specifically did not report fine content of substrate for spikedace. However, glide habitats were reported to be composed of a mean of about 30 percent sand (fine) substrate and run habitats were characterized by a mean of about 25 percent sand or fines. Although not defined at that time, a re-calculated average fine content of "glide-run" habitats from Rinne and Stefferud (1996) is about 29 percent. Neary et al. (1996) documented that spikedace numbers increased almost three-fold (18 to 52 individuals) when the fine component of the substrate decreased from about 27 percent to 7 percent.

In the upper Gila River, Rinne and Deason (2000) reported that spikedace were most commonly found over sand-gravel substrates. Sand content or fine levels averaged 12 percent where spikedace were present to 5 percent where absent. Further, the sand content in all glide-run habitats sampled in the Verde and Gila Rivers in 2000 was 18 and 20 percent, respectively. Sand substrate in glide-run habitats in the upper Verde River declined from 50 percent (range = 50–53, n = 7) in 1996 to 38 percent (range 10–77, n = 7) in 2000 (Rinne, unpublished data). Rinne (1991) reported spikedace located over sand substrates, but use of these habitats varied by stream.

Loach minnow was captured almost exclusively (90%+) in high-gradient riffle habitats in the upper Gila. In these habitats, substrates had a mean of 8.3 percent fines in 1999 and 8.8 percent in 2000. Of 49 high-gradient riffles sampled in the upper Gila in 1999 and 2000, fines averaged only 6 percent of the substrate composition. Based on substrate indexes, Rinne (1989) documented that loach minnow minimally occupied habitats over sand substrates.

Finally, if one compares the abundance of spikedace and loach minnow in respective aquatic macro-habitats where captured, there are no obvious, consistent, indirect trends of relationships between percentage fines in substrates and densities of fish (Tables 1 and 2).

Discussion

Spikedace

Spikedace has declined markedly between 1994 and 1999 in the upper Verde River (Rinne 1999b). By comparison, the species is yet present and locally abundant in the upper Gila River (Rinne et al. 2001; Rinne in press). Fine sediment in optimum glide-run habitats is near identical (18 and 20%) in the two rivers (Rinne and Deason 2000). Further, there is no apparent indirect trend in abundance of spikedace relative to fine sediment concentration (Table 1). Livestock grazing has been removed from the upper Verde River

Table 1. Relationship of abundance of spikedace in the upper Gila River, 1999–2000, relative to percentage fine content of substrate. Absence of spikedace is calculated only from glide-run habitat type.

	Sample size	Abundance category	Range of fine content	Mean fine content
1999				
	32	0	0–96	24
	16	< 5	0–88	28
	11	5–9	4–88	20
	18	> 10	0–46	17
2000				
	30	0	0–60	–
	13	< 5	0–41	13
	8	5–9	0–29	1
	2	> 10	15–41	28

Table 2. Relationship of abundance of loach minnow in the upper Gila River, 1999–2000, relative to percentage fine content of substrate. Absence of loach minnow is calculated only from high-gradient riffle habitat type.

	Sample size	Abundance category	Range of fine content	Mean fine content
1999				
	25	0	0–28	3.0
	24	< 5	0–64	9.5
	9	5–9	0–38	12.0
	5	> 10	0–18	6.0
2000				
	8	0	0–26	6.3
	13	< 5	0–65	4.0
	3	5–9	0	10.0
	2	> 10	0	6.3

corridor since 1997; it is present in the Gila–Cliff Valley reach of the upper Gila River. This reach of the upper Gila has been reported as the area of greatest concentration of the species for almost 2 decades (Propst et al. 1986; Rinne et al. 2001).

Limited data on the upper Verde (Neary et al. 1996) suggest that spikedace abundance increased markedly relative to marked decrease in sand substrates. Nevertheless, substantial numbers of spikedace were present and in spawning aggregations in aquatic habitats containing substrate fine levels of up to 27 percent. By comparison, in the upper Gila in 1999–2000 spikedace were collected over substrates with fines averaging 21 percent (Table 1).

Loach Minnow

Although historically present in the Verde River, I only have data on loach minnow presence and abundance for the upper Gila River, 1999–2000. This species almost exclusively occupied high-gradient riffles (slope > 1.0%, mean 2.2%) characterized by gravel-pebble (70%) and cobble (14%) substrates. The almost complete lack of this habitat type in the upper Verde in 1999 (Rinne et al. 2001) must certainly reduce the probability of loach persistence in the upper Verde River. It may further suggest why historically this species was not collected in the extreme upper Verde River.

In both 1999 and 2000, fines in high-gradient riffle habitats averaged less than 9 percent. A mean of only 6 percent fines was calculated for all high-gradient riffle habitats in the upper Gila. As recorded for spikedace, there was no inverse trend in loach minnow abundance and fine sediment concentration (Table 2). The flushing process caused by higher velocities (mean 68 cm/sec) in high-gradient riffles moves fines through these habitats naturally, resulting in low substrate fines levels. Propst et al. (1988) reported that loach minnow occupied the upper erosional, less consolidated portions rather than the lower depositional, more sedimented portions of these habitats. Although not specifically recorded, I have also noted this linear distribution of abundance of loach minnow in high-gradient riffles.

Similar to spikedace, loach minnow spawning is likely the critical life history stage that is affected by fines. Resting cover and the effects of fines on food supply are likely of secondary limiting nature. Propst et al. (1988) documented that deposition of eggs in riffle habitats occurs at the downstream undersurfaces of cobble and boulder substrate components. In the event of excessive fines in the bedload these areas would conceivably become filled because of extant hydrologic function which would deposit these fine sediment materials in these areas of lower velocity. However, the fines (ca. 9%) recorded in high-gradient riffles in the upper Gila do not appear to approach an excessive level. As with spikedace, specific study is needed to substantiate this contention.

Conclusions

The data suggest that fines cannot be unequivocally identified as negative impacts on spikedace or loach minnow in these two desert rivers. Further, neither can grazing-generated fines be implicated as a major limiting factor, because this land use is present at moderate to high levels in the Gila–Cliff Valley where both species are present. Also, it is here that loach minnow and spikedace have persisted through time (Propst et al. 1986, 1988; Sublette et al. 1990) and are currently present and locally abundant (Rinne et al. 2001). Additional detailed, controlled studies are needed to unequivocally document the threshold of the effect of sediment fines on spikedace and loach minnow sustainability.

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LABOR DAY STORM OF 1970 REVISITED 30 YEARS LATER

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The 1970 Labor Day storm caused more loss of human life than any other storm in Arizona's recent history. Many dwellings, roads, bridges, and other structures were also damaged or destroyed by the record flooding. Most of the widespread and unprecedented human and economic losses occurred in central and northeastern Arizona, with other losses reported in southeastern Utah and southwestern Colorado (Roeske 1971; Thorud and Ffolliott 1973). At the request of Governor Jack Williams, President Nixon declared the flood-damaged areas a major disaster, permitting political jurisdictions and, in a restricted sense, private entities to be reimbursed by the federal government for eligible expenditures made as a result of the storm. It is difficult to assess the total dollar cost of the storm, although it has been estimated that initial expenditures to immediately repair storm damages to infrastructures approximated $25–35 million in terms of current dollars.

Preliminary reports on the storm and its effects were presented at the 1971 Arizona Watershed Symposium (Thorud and Ffolliott 1971) and the 1972 Western Snow Conference (Thorud and Ffolliott 1972). A more comprehensive report on the 1970 Labor Day storm and associated flooding was prepared at the request of the Arizona Water Resources Committee, a private nonprofit organization formed (but later disbanded) to promote the development of Arizona's water resources with particular emphasis on water yields from watershed lands (Thorud and Ffolliott 1973). Meteorologic and hydrologic features of the storm, the resulting upland watershed damages, relationships to land management practices, and the degree to which upland watershed damages have been mitigated in the 30 years since the storm are reviewed in this paper.

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The Storm and Its Consequences

The 1970 Labor Day storm caused rainfall of varying amounts at different locations in Arizona beginning on September 3 and ending mostly on September 6. Skies cleared over much of the state by late afternoon on September 7, and relatively little rainfall was reported for that observational day. Flooding began near the border with Mexico on September 4, occurred in the Mogollon Rim area in central Arizona and westward on September 5, and continued along the Little Colorado River basin and in the Tucson vicinity on September 6. High peak streamflows and flooding were observed at various locations, but the most disastrous flooding and loss of human life occurred on September 5.

Meteorological Event

The conditions that led to the 1970 Labor Day storm developed initially with a northward advance of moist, unstable air from the eastern Pacific Ocean and Gulf of California that was associated with tropical storm Norma (National Oceanic and Atmospheric Administration 1970; Thorud and Ffolliott 1973). Following this air mass invasion, the triggering mechanisms that contributed to heavy rainfall in the state included an orographic uplift associated with strong southerly winds in the lower atmosphere, the invasion of an unusually intense late-summer cold air mass from the Pacific Northwest with its associated frontal activity, and daytime convective heating over the desert valleys.

Rainfall totals of 5 inches or more were associated with the mountainous terrain of the Mogollon Rim northeast of Payson, the Sierra Ancha Mountains southeast of Payson, the Mazatzal Mountains south and southwest of Payson, the Bradshaw Mountains east of Prescott, the high country south of Flagstaff, the Santa Catalina Mountains northeast of Tucson, and the Baboquivari Mountains and Kitt Peak southwest of Tucson

(Thorud and Ffolliott 1973). New precipitation records for a 24-hour observational day were established at many National Weather Service stations. Perhaps the most spectacular record was established at Workman Creek in the Sierra Ancha Mountains, where 11.4 inches of rainfall were recorded in an official rain gauge between 2200 hours on September 4 and 2000 hours on September 5, a new record for the state (National Oceanic and Atmospheric Administration 1970; Kangieser 1972). The previous official National Weather Service record for a 24-hour observational day was 6 inches, recorded at Crown King on December 19, 1967. Rainfall intensities of greater than 3 inches in 4 hours were reported for several stations during the storm period (Roeske 1971). Rainfall intensities of these magnitudes easily exceed the infiltration rates on many watershed lands with shallow soil storage over bedrock, facilitating large amounts of surface runoff and high peak streamflows.

A 24-hour rainfall total of 5–6 inches is a 100-year event at many locations in central Arizona (U.S. Weather Bureau 1967). This amount of rainfall was equaled or exceeded at several of these locations during the storm (Thorud and Ffolliott 1973). Such a storm does not necessarily occur at 100-year intervals, however. The same event could occur 2 or more years in a row at the same location, although such a sequence is unlikely.

Hydrologic Event

High peak streamflows and flooding events occurred as a result of the storm. Peak discharges of several streams in central Arizona, where the most damaging flooding occurred, exceeded the 20- to 25-year flood (Thorud and Ffolliott 1973). The return period was much higher on small watersheds of 25 square miles or less. As expected, much of the flooding was associated with areas receiving the higher rainfall amounts and intensities. Record peak discharges and flood stages were measured at selected stations (Roeske 1971). At least 30 USDI Geological Survey gauging stations in the Gila River Basin measured record peak streamflows.

The highly used recreation sites near Kohl's Ranch on Tonto Creek were severely damaged by flooding, and sadly, it was here that more human lives were lost than at any other area (Elson 1971; Thorud and Ffolliott 1973). The hydrologic event here was, therefore, of special interest. An estimated peak streamflow of about 18,400 cubic feet per second (cfs) occurred on Tonto Creek at Kohl's Ranch on September 5. This flow, combined with the high flows from two tributary streams (Chistopher and Haigler Creeks), resulted in a peak streamflow of 38,000 cfs at Tonto Creek near Gisela on September 5. Studies of recurrence intervals have suggested that the peak streamflow of Tonto Creek near Gisela was a 10-year event (Thorud and Ffolliott 1973). This means that, on the average, a flood of this magnitude will be equaled or exceeded once in 10 years over the long run. Or, in other terms, there is a 10 percent chance of a flood of this magnitude being equaled or exceeded in any given year. However, the situation was compounded by the frequent breaching of debris dams during the flood.

Reservoirs on rivers and streams helped to reduce the damage potential of the flood flows. Roosevelt Reservoir on the Salt River stored all of the flow from the Tonto Creek Basin, and Horseshoe Reservoir on the Verde River to the west absorbed the flows from its upstream tributaries (Roeske 1971). The storage in these reservoirs helped to prevent significant damage to Phoenix and other downstream population centers. Some of the small reservoirs (recreation lakes) on upland watersheds had available storage capacity at the time of the storm, and therefore stored water to also prevent or reduce destruction downstream (Thorud and Ffolliott 1973). However, some upland reservoirs received an influx of floating woody debris that was considered to be potentially hazardous, because it could jam spillways during future runoff events and contribute to dam failures.

Upland Watershed Damages

Many stream channels on upland watersheds were detrimentally altered as a result of flooding during the storm. Types of damages included accumulations of uprooted trees and other materials in debris dams at restriction points in the channels, deposition of boulder fields in the channels, channel scouring (to bedrock in some channels), and bank cutting (Morrison 1970; Willams and Russell 1970; Elson 1971; Arnolt 1972; Thorud and Ffolliott 1973). Debris dams resulted from high streamflows carrying uprooted trees and other vegetative debris and rocks. This churning mass of material tended to lodge and form the dams in narrow places, on sharp curves, in stands of trees, and around bridges and culverts. These channel diversions resulted in a head of water being built up behind the dams; these dams often breached, sending a surge of water, timber, and rocks downstream in flood waves to the next restriction point where the process might be repeated. There was a concern that the debris left in the channels following the storm might again plug channels, culverts, and bridges, or divert subsequent spring snowmelt streamflows into the unstable banks, causing more erosion. As a consequence, riparian and channel restoration efforts were undertaken on some of the stream systems.

Massive boulder fields were deposited at various locations in some of the channels. Some deposits were 10–30 feet deep, extending the width of the channels, and up to 2 miles in length. Rock size varied, but the larger boulders were 6 cubic yards in volume and weighed up to 50 tons. Streams flowed through or under the boulder fields in some places, while water was diverted laterally toward the banks elsewhere. This latter situation was undesirable because it could lead to further bank erosion and attendant soil loss and more trees could be undermined and dropped into the channels. Future channel damage could likely be caused by rock piles considered unstable and subject to movement during subsequent high streamflows. Channel scour to bedrock occurred in some locations; a tributary of Tonto Creek was virtually swept clear of loose sediments above bedrock for much of its length. Creation of vertical stream banks was another result of flood flows during the 1970 Labor Day storm, particularly where debris deposits caused streamflows to be diverted against the banks, at curves in channels, and where channel scouring was deep. Some of the vertical banks were unstable and constituted a potential source of sediment and timber debris due to sloughing during subsequent high flows. The hazard of unstable banks was accentuated where trees and large boulders were precariously suspended at high locations on or near vertical stream bank faces.

Damage to Fishery Resources

Damage to fishery resources was extensive throughout the flood area (Arnolt 1972; Thorud and Ffolliott 1973). Streams were sometimes split into multiple channels by rock piles, often with insufficient flow to support fish populations. Other conditions detrimental to the fishery resources (resulting from the flood and difficult to correct) were created by channel scouring to bedrock, filling of pools with boulders, sand, and silt, and the diversion of channels. The loss of streambank (riparian) vegetation which ordinarily shades the stream resulted in water temperatures too high for trout populations in some instances.

Relationship to Land Management Practices

Hydrologic relationships between land management practices on upland watersheds and overwhelming meteorologic events such as occurred with the 1970 Labor Day storm are difficult to isolate and quantify (Thorud and Ffolliott 1973). If a large rainfall event occurs in a short period of time, flooding might occur regardless of the management practice. However, if a watershed has been largely cleared of vegetation (for whatever reason) and has bare soil exposed to the elements, the rates of surface runoff, erosion, and sedimentation can be higher than would be expected if a protective vegetative cover was on the watershed. The hydrologic response of the Beaver Creek watersheds in north-central Arizona to the Storm (Baker et al. 1971; Brown et al. 1974; Clary et al. 1974) provided insight to this phenomenon. Total runoff, peak streamflow, and total sediment yield associated with the storm were higher on a watershed in the pinyon-juniper woodlands that had been cleared (cabled) of overstory vegetation 6 years earlier than would otherwise be expected. Total runoff and peak streamflow from the storm were also higher on watersheds in the ponderosa pine forests that had been either totally clearcut 3 years earlier, thinned 1 year earlier, or partially clearcut in strips 3 years earlier. Total sediment yield was highest on the watershed that had been totally clearcut. Although strictly limited to conditions at Beaver Creek, the analysis by Baker et al. (1971) was helpful in attempting to understand the relation of the 1970 Labor Day storm to some of the land management practices that had been implemented on the upland watersheds in Arizona at the time of the storm.

Restoration Activities

Restoration activities on larger, often perennial stream systems whose natural hydrologic functioning was damaged by the storm included corrective actions taken to mitigate the effects of boulder accumulations, timber-related debris, vertical stream banks, channel scour, sand and silt deposits, stream channel diversions, road and trail damage, loss of streamside vegetation, and bank-hanger and pedestaled trees (Arnolt 1972). Boulders were pushed against cutbanks to form riprap to stabilize the banks. Channels were cut through boulder fields to divert water through the boulders and away from stream banks. Margins of boulder accumulations were stabilized by leveling them out with dozers. Some of the timber-related debris piles were burned in place. Non-timber debris was removed from trees where it had lodged and was piled and burned. Logs were skidded onto high ground, decked, and either left there or loaded onto trucks and hauled to disposal sites. Bank-hanger and pedestaled trees were removed as a preventative measure. Large and long logs were placed at the bases of vertical stream banks; the tops of the banks sloughed off on top of many of them. Vertical stream banks were sloped with dozers to a stable angle of repose and seeded with grass species or planted with willows, alders, and other tree species. Backhoes and dozers were used to consolidate multiple channels created by the storm into one flow pattern by lowering one channel below the others.

Plantings of willow, alders, and other tree species restored much of the lost streamside vegetation. Live trees that were undercut by the flood were removed from stream channels and stream banks by cutting. Pedestaled trees that were situated part way up the bank where poor footing existed were felled by blasting with high explosives. The most extensive of these restoration activities occurred on the Tonto National Forest, including Tonto Creek from its headwaters to Bear Flat, the East Verde from its headwaters to East Verde Park, and Christopher Creek from its headwaters to the Ellinwood Crossing. On-the-ground surveys made in January of 1971 determined that approximately 13.7 miles of stream required restoration activities of some kind because of the potential hazard to life and property. All of this work was performed within the water influence zones of the various streams.

Sites Revisited

Insights on the degree to which upland watershed damages have been mitigated in the 30 years since the 1970 Labor Day storm have been derived from limited and largely qualitative observations from the Kohl's Ranch–Tonto Creek area and the Beaver Creek watersheds, and analysis of paired 35 mm color photographs (slides) from selected locations. One set of photographs was taken immediately after or within 1 year of the flooding. The second set of photographs was taken at these locations in October of 2000.

Channel Characteristics

Trees and other vegetative materials in the debris dams formed as the result of flooding have largely decomposed. Only a few larger tree parts and some of the larger sediments that accumulated behind the debris dams remain visible. Channel restoration on some of the streams after the flood has further obliterated signs of the dams. Most of the restoration took place on channel sites where high seasonal streamflow is expected and around bridges and culverts. Large boulders in channels remain where they were deposited in the flood. However, smaller sediments have washed away in the intervening time. It is not possible to relate subsequent streamflow volumes to the movement of these smaller sediments, either because the streams are ungauged or because gauging was discontinued at some point since the flood, precluding a complete record of streamflows. Vegetation has become established near the stream banks in some of the original boulder fields, especially on sites with low streamflow regimes. Evidence of channel scour remains at some locations, although accumulations of sediments are found in some instances. Vertical stream banks created by the flood also remain along channels that carried the larger flood flows. Some of these banks have sloughed off into the stream channels.

Status of Fishery Resources

Fishery resources have responded favorably to the restoration work on Tonto Creek and other perennial stream systems in the flood area. Creation of artificial pools and riffles and the reestablishment of streamside vegetation have especially benefited trout populations in these streams. The Tonto Creek Hatchery was rebuilt by the Arizona Game and Fish Department after being destroyed when a dam broke and sent a 35-foot wall of water down on the hatchery during the flood.

Effectiveness of Restoration Activities

Observations made in October of 2000 on the effectiveness of the restoration activities suggest that the hydrologic functioning of the treated perennial streams has been largely restored. Bank erosion is not excessive, streamflow response to precipitation appears relatively slow, and baseflow is sustained between storms. There are accumulations of sediments on many of the scoured channel bottoms. Streamside vegetation consisting of small trees, shrubs, and herbaceous plants has also been reestablished artificially and naturally along many of the streams to stabilize the banks and maintain water temperatures in a range favorable to the stocked trout populations. Although a rigorous benefit-cost analysis of the restoration activities has not been made, the investment made 30 years ago in restoring the natural hydrologic functioning of streams following the 1970 Labor Day storm seems justified.

Summary

The 1970 Labor Day storm caused more loss of human life than any other storm in Arizona's recent history. Many dwellings, roads, bridges, and other structures were also damaged or destroyed by the record flooding. Much of the widespread and unprecedented losses occurred in central and northeastern Arizona. Stream channels on many upland watersheds were detrimentally altered by breached debris dams of uprooted trees and other materials, depositions of boulder fields, channel scouring, and bank cutting. Some boulder fields were 10–30 feet deep, extending the width of the channel, and up to 2 miles in length. Streams were sometimes split into multiple channels by rock piles, often with insufficient flow to support fish populations. Damage to fishery resources was extensive. Restoration activities undertaken on the larger streams included corrective actions taken to mitigate the effects of boulder accumulations, timber-related debris, vertical stream banks, channel scour, sand and silt deposits, stream channel diversions, road and trail damage, loss of streamside vegetation, and bank-hanger and pedestaled trees. The most extensive restoration activities occurred on Tonto Creek, the East Verde River, and Christopher Creek.

Insights on the degree of damage mitigation after 30 years are derived from limited and largely qualitative observations from the Tonto Creek and Beaver Creek watersheds areas. Trees and other vegetative materials in the debris dams have largely decomposed. Only a few larger tree parts and some of the larger sediment accumulations remain visible. Channel restoration on some of the streams after the flood has further obliterated signs of the dams. Vegetation has become established in some of the boulder fields and many of the scoured areas have new accumulations of sediment. Creation of pools and riffles and reestablishment of streamside vegetation have benefited trout populations. Observations on the effectiveness of restoration activities 30 years after the 1970 Labor Day storm suggest that the hydrologic functioning of many of the treated streams has been largely restored. Bank erosion has not been excessive, streamflow response to precipitation appears relatively slow, and baseflow on perennial streams is largely sustained between storms. Streamside vegetation of small trees, shrubs, and herbaceous plants has been reestablished to stabilize most banks and help to maintain water temperatures in a range favorable to the trout populations.

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