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TABLE OF CONTENTS

INTRODUCTION

The spring meeting of the Hydrology Section of the Arizona-Nevada Academy of Sciences took place in Tucson, Arizona on April 20, 1996. We want to extend our thanks to Ms. Louella Holter, Bilby Research Center, Northern Arizona University, for her editorial assistance on this publication; and to Don W. Young, the 1996 Hydrology Section Chairman.

POTENTIALLY POLLUTING ACTIVITIES AND THE CONTROL OF ENVIRONMENTAL RISK: UNDERGROUND STORAGE TANKS AND AQUIFER PROTECTION (PIMA COUNTY)

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Early national concern over water pollution focused on the contamination of surface water, with one of the first laws being enacted in 1899 (the Rivers and Harbors Act), followed by the Federal Water Pollution Control Act of 1948 (62 Stat. 1155, 33 U.S.C. §§ 466 to 466g). Little progress was made until, in 1972, the United States Legislature passed the Clean Water Act (CWA; 33 U.S.C. §1251 et seq.) regulating "point-source" discharges (such as from factories) into navigable waters (surface water) (Goplerud 1995). However, there has been considerable evidence that groundwater is also at risk from a variety of sources such as industrial waste burial, fertilizers and pesticides used in agriculture, chemicals leeching out of landfills, mining by-products, and leaking underground storage tanks. Concern over land and groundwater contamination led Congress to enact a series of remedial statutes including the Safe Drinking Water Act in 1974, setting limits on contaminants allowed in drinking water (SDWA; 42 U.S.C. §300f et seq.); the Resource Conservation and Recovery Act in 1976, regulating the production, transportation and storage of hazardous waste (RCRA; 42 U.S.C. §6901 et seq.); the Comprehensive Environmental Response, Compensation and Liability Act in 1980, authorizing the cleanup of abandoned hazardous waste sites (CERCLA; 42 U.S.C. §9601 et seq.); and Subtitle I added to RCRA in 1984, regulating underground storage tanks (42 U.S.C. §6991 et seq.).

Underground storage tanks (USTs) typically are used to store gasoline, diesel fuel, waste oil, aviation fuels, kerosene, cleaning solvents, and other volatile materials at automotive stations and large municipal fleet service yards. These substances contain chemicals that are toxic and carcinogenic when released into groundwater; when leaked, they produce vapors that migrate through soil and may collect in sewer lines and basements, causing fires and explosions, as well as poisoning human beings (ADEQ 1988a, 1989; EPA 1995).

In the 1950s and 1960s, the United States experienced a boom in the installation of USTs; by the early 1980s, many of these now old tanks were corroded and leaking, causing a significant increase in the number of serious contamination incidents. During a 1983 Senate hearing, Jack Ravan, the assistant administrator for water for the Environmental Protection Agency, estimated that 75,000–100,000 USTs were leaking 11 million gallons of gasoline annually into groundwater (Ninety-eighth Congress 1983; Hayward 1994). One gallon of gasoline can potentially contaminate up to 750,000 gallons of water (Hayward 1994). In 1984, in response to growing evidence of groundwater and soil contamination resulting from leaking USTs, Congress added Subtitle I to the RCRA. It directed the Environmental Protection Agency, which had been created by Congress with the enactment of the Clean Air Act of 1970 (42 U.S.C. §7401 et seq.), to develop a program to resolve the UST problem. In 1986, the Environmental Protection Agency (EPA) reported the results of a two-year UST study in which 433 tanks were tested: 35 percent were found to be leaking and 55 percent of the leaking tanks were involved in leakage of motor fuel into groundwater (Gauthier 1990). In 1991, David W. Ziegle, the EPA acting director of the Office of Underground Storage Tanks, estimated that there were more than 2 million USTs then in existence at more than 750,000 facilities in the U.S., with 100,000 confirmed leaks and an additional 300,000 confirmations anticipated in the near future (Trial 1991). Currently, it is estimated that more than 25 percent of USTs are leaking (Kohout 1995).

Because of the great expense and difficulty of remediation (remediation of a single site can cost in excess of $1 million when groundwater is involved), much of the RCRA Subtitle I statutes and

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pursuant EPA and state regulatory framework is focused on prevention. The RCRA statutes define USTs and determine which types of USTs are required to comply with standards; require owners of USTs to notify states of the existence of USTs on their property; require new tanks installed after 1988 to meet design, construction and installation standards; set standards for leak, cleanup and tank closures; and provide the EPA authority to issue administrative orders or pursue federal civil actions to enforce these statutes (42 U.S.C. § 6991-6991i). The EPA chose to design a set of rules defining a minimum standard for protecting the environment while allowing states and counties the freedom to pass more stringent regulations (Taylor 1991) and has statutory authority to delegate regulatory authority to the states by approving state UST programs (42 U.S.C. 6991c.) In Arizona, the threat posed to groundwater and soil by leaking USTs is considered to be of statewide concern and counties are prohibited from enacting regulation (A.R.S. §49-1010). However, Arizona has also chosen to require that state regulations be no more stringent than federal regulations (Chap 231 H 2196 1995 Legislative Enactment).

The Arizona Department of Environmental Quality (ADEQ) was established by the Arizona Environmental Quality Act of 1986 and began operations July 1, 1987. Prior to this, environmental concerns were handled through the Department of Health Services. In 1988, the EPA assisted in funding the ADEQ to develop a state program to enforce federal UST regulations through the Office of Water Quality (A.R.S. §49-1001 et seq.). The ADEQ's Underground Storage Tank Section was established in September 1990, with 60 full-time employees (ADEQ 1991).

In Pima County, as of October 20, 1995, there were 1,629 tanks in use at 546 facilities registered with the Arizona Department of Environmental Quality; almost all of the tanks were in the metro Tucson area. There were 618 leaks reported at 534 facilities between May 1985 and October 1995; as of October 1995, 399 cases were still being remediated. Forty-one of these cases impacted groundwater and 241 cases were confined to soil contamination. Contaminants beneath the downtown Tucson area, leaking over the years from aging USTs, have seeped into the perched groundwater; there is evidence of gasoline in the vadose zone, and halogenated volatile organic compounds have been discovered in the soil and shallow groundwater. Diesel contamination has been found in one of the city's production wells southeast of the downtown area, Well B-78A, which has been closed (ADEQ 1995b).

The UST Program and Protecting Groundwater in Arizona

Groundwater constitutes over 60 percent of Arizona's water supply (as of 1990; ADWR 1990). At present, the sole source of drinking water for the city of Tucson is groundwater obtained from a deep regional aquifer which underlies the Tucson Basin. Although in most areas the water table now lies 50 or more feet below the surface, dropping water levels from overpumping have left shallower pockets of water perched above the deep aquifer in many areas. Such perched water may be totally isolated from the deeper aquifer or there may be some leakages. Contamination from the surface can directly reach water in the deep aquifer by percolating through the layers of soil, sand, and gravel above it (the vadose zone) or it can reach areas of perched water and may seep into the aquifer if there are any connections. In addition, wells drilled into the deep aquifer and passing through areas of contaminated perched water may provide a conduit for the contaminated water to "cascade" along the well shaft and contaminate the deeper water (Gregory Hess, personal communication 1996).

The federal and state regulations comprising the UST program fall into two categories: regulations controlling the activity (storage of petroleum products in tanks) to minimize contamination from the activity; and regulations imposing liability for the costs of damages, including stipulating actions to be undertaken in case of accidental contamination. The object is to induce tank owners to exercise care in the operation and maintenance of their tanks. The regulations impose two types of costs on a tank owner/operator: the cost of preventative measures and, to some degree, the cost of repairing damage to human health and the environment from accidental spills.

Technical Requirements for USTs

The cost of preventative measures is borne directly by businesses that use underground storage tanks as part of their operations. To some degree, this cost may be passed on to the consumers of a business's product in the form of higher prices, with these prices incorporating the cost of reducing the probability of contamination incidents.

Technical requirements apply to both new and existing tanks. Areas addressed are tank construc-

tion, piping, spill and overflow prevention devices, leak detection, rules for installation and closures, upgrade requirements, and rules for operation and maintenance. The three main safeguards against leaks are leak detection, spill and overfill devices, and corrosion protection (A.R.S. §49-1009).

Tanks and piping installed after December 22, 1988 must comply with all safeguards at installation. Existing tanks and piping, installed before December 22, 1988, must comply with leak detection regulations no later than December 1993; spill, overfill, and corrosion protection compliance must be accomplished no later than December 22, 1998.

Postregulation USTs with double walls and leak monitoring and tank gauging systems cost two to three times the price of the preregulated UST. Nationally, it has been reported that the majority of the owners of the leaking USTs have opted to buy aboveground storage tanks (ASTs) because of the extensive monitoring procedures and insurance requirements for USTs. ASTs do not fall under the same safety regulations as USTs but they also pose certain problems in regard to esthetics, corrosion protection, fire concerns, vapor emissions, thermal expansion and contraction, and vulnerability to vandalism (Safety & Health 1994; Civil Engineering 1994). Figures for AST installations in Arizona have not yet been obtained.

In addition to regulating the design and construction of USTs, the law addresses tank installation and closure, to minimize contamination during these procedures, as well as operation and maintenance (A.R.S. §§49-1002, 1003, 1004, 1008, 1009).

Inspectors from the ADEQ's UST Inspections and Compliance Unit are authorized to carry out compliance checks to encourage owners/operators to adhere to UST regulations. Sites to be inspected are prioritized by UST age, capacity, number of USTs at the site, and content (hazardous substances or not). Underground Storage Tank News (Winter 1994/1995) reported approximately 24,000 active USTs in the state (1,639 in Pima County) at 3,900 UST facilities (546 in Pima) (Christman 1995; UST Section Database). Currently, 81 percent of all tanks in operation in Pima County were installed prior to 1988 (UST Section Database). The Inspections and Compliance Unit has 21 case managers (Pima Association of Governments 1994).

Liability When Damage Has Occurred

Leakage from underground storage tanks first impacts the surrounding soil. Once in the soil, the contaminants can percolate through the soil until they intersect with water. Rain passing through the soil can also capture the contaminants and carry them faster into groundwater supplies. Contaminants in the soil can potentially cause damage to human health depending on exposure, as well as to animals and vegetation. The threat to groundwater, and potentially to drinking water, depends on how deep the water table is, where drinking water wells are located, how much rain there is, and how quickly the contaminants can penetrate the soil. If drinking water becomes contaminated from spills or leakages, public water systems, which are required under the SDWA to test and treat water, are burdened with increasing treatment costs. The probability of health damages from ingesting contaminated water can increase if testing is inadequate and the SDWA regulations are not enforced. The UST statutes require owners of USTs to clean up any contamination from leaks or spills in order to reduce the potential water treatment cost increases and cost of expected health damages.

State and federal regulations require extensive procedures for reporting leaks and spills as well as for undertaking corrective actions as quickly as possible in order to minimize impact. The average cleanup cost for a site contaminated by a leaking UST in 1993–1994 was estimated to be approximately $225,000 (Ameden 1995). Cleanup costs increase as contamination spreads into the surrounding soil, and costs skyrocket if groundwater is contaminated (EPA estimates in 1991 ranged between $50,000 and $1 million). Owners/operators are required to notify ADEQ within 24 hours of confirming a release and to begin cleanup within that time as well. Petroleum products must be removed from the UST system to prevent further release. Next, the owner must ascertain the extent to which the petroleum has spread, begin collection, and file a progress report no more than 20 days after the release report. Owners must investigate the extent of soil and groundwater contamination, report this to ADEQ, and submit a corrective action plan for meeting environmental standards (EPA 1995).

When Damage Has Occurred, How Clean Is Clean?

As soil and groundwater is cleaned, a major issue is how clean is clean? The EPA sets standards for the level of contaminants acceptable in soil and groundwater based on risks to human health. Originally, the statutes required that the same

standard be applied to all sites being remediated: the higher the level of cleanliness required, the greater the cost of achieving it. However, funds for cleanup have been limited. Discontent in the regulated community with the burdensome costs of remediation prompted the ADEQ to develop more flexible and cost-effective guidelines for determining the level of cleanliness required for soil at any site. A similar set of flexible guidelines for determining standards for groundwater cleanup may also be developed in the near future.

In August 1994, the ADEQ Cleanup Policy Task Force began developing a general soil remediation policy applicable to all ADEQ programs, intended to be as flexible as possible to reduce remediation costs. On January 3, 1995, the ADEQ adopted its interim policy on soil remediation based on task force recommendations. Laws passed in the 1995 legislative session mandated the development of risk-based, consistent soil remediation rules. The ADEQ adopted emergency interim rules on December 15, 1995 (§49-151,152; §49-1026) which were sent for approval to the secretary of state. Legislation requires that final rules be adopted by August 1, 1996 (ADEQ 1995c).

The new soil remediation rules, if approved, would allow sites to be classified according to future use as either industrial or residential. Once classified, owners can choose to clean the site to predetermined contaminant concentration levels; these health-based guidance levels, specified by the Arizona Department of Health Services and based on the toxicological characteristics of each specific substance, bear no relationship to the characteristics of a given site. Residential standards (permissible concentration levels) are stricter and more protective than industrial standards; if there is potential for groundwater contamination at the site, standards stricter than the residential ones could be imposed, especially for benzene, toluene, ethylbenzene, and xylenes. Alternatively, owners can have a site-specific risk assessment study carried out to determine standards of cleanup for either residential or industrial use; such assessments include consideration of potential harm to groundwater (McAllister 1996). Copeland and colleagues profiled three cases involving the remediation of petroleum-contaminated sites in Arizona and California where site-specific risk assessments were carried out: two of the cases involved remediation of sites for industrial uses, and the last case involved property that a school planned to purchase. In the first case, the risk assessment study concluded that the contaminated soil did not threaten health or groundwater and therefore no remediation was necessary; the cost of the study was $11,000 and the cost of carrying out remediation to predetermined levels would have been $120,000, with a savings of $109,000. In the second case, the risk-assessment study also found that no remediation was necessary; the study cost $15,000 and remediation would have cost $500,000 with a savings of $485,000. In the case of property that was to be purchased by a school, the findings were that remediation was necessary and the cost was $125,000; however, the $13,000 risk assessment study did result in reductions in remediation costs (Copeland et al. 1995/96).

Who Pays? Financial Responsibility and the State Assurance Fund

Subtitle I of the RCRA establishes that UST owners are to be held liable both for remediation costs in the event of a spill, leak, or overflow and for third party bodily injury and property damages (40 C.F.R. Part 280 Subpart H, 1988, amended Dec. 23, 1991; § 280.98 b). To insure that owners/operators can meet this double requirement to some degree, they must demonstrate financial responsibility by providing financial assurances of these funds. (All owners had to meet these requirements by Feb. 18, 1994.) Specifically, marketing facilities or facilities with a throughput of more than 10,000 gallons are required to have some way of providing $1 million per release; smaller facilities (throughput of less than 10,000 gallons) must be able to provide $500,000 per release (Alspach 1990; UST News Fall 1992; 40 C.F.R. § 280.93). Owners of 1–100 USTs would be responsible for no more than $1,000,000 per year no matter how many releases occurred; owners of more than 100 USTs would be annually liable for no more than $2,000,000. In cases where the state must intervene to remediate a site, it is empowered to sue the owner for the costs incurred (when a responsible party is available).

The Arizona UST program is funded from both state and federal sources. In 1986, amendments to CERCLA established the federal UST Trust Fund; revenue is generated by a 1 cent per gallon federal tax on petroleum products, and grants from this fund are then distributed to the states (Douglas Wheeler, personal communication 1995; Taylor 1991). The EPA grants are deposited in Arizona's Leaking Underground Storage Tank (LUST) trust fund to be used by Arizona to assist with the cost of remediating both abandoned sites (often belonging to businesses that failed during the oil

crises or ensuing recessions) and sites where the owner is insolvent.

In 1989, concerned that meeting federal financial responsibility requirements would be a serious hardship for smaller operations, the ADEQ and state legislature began to develop state sources of funding (ADEQ 1989). The resulting 1990 amendments to the Arizona Underground Storage Tank Act (H.B. 2011) created the Arizona Underground Storage Tank Revolving Fund which is divided into three separate accounts (A.R.S. § 49-1015): the regulatory account (A.R.S. § 49-1015 A-D), the state assurance fund (SAF) (A.R.S. § 49-1051 to § 49-1056) and the grant account (A.R.S. § 49-1071 to § 49-1073). By mid-1992, rules governing financial responsibility and the SAF were adopted (A.A.C. R18-12-300 to R18-12-321; A.A.C. R18-12-601 to R18-12-610; ADEQ 1993) and the SAF program began processing applications September 22, 1992.

Although the state can access the SAF for state-initiated cleanups and legal actions, the fund is to be used primarily to assist owners with the costs of mitigating public health and environmental damages resulting from leakages (for costs incurred after Sept. 15, 1989) and to provide a means of partially meeting the financial responsibility provisions. Revenue for the SAF is obtained from a 1 cent per gallon state excise tax on gas and petroleum products that flow through regulated tanks in a given calendar year (A.R.S. § 49-1031 to § 49-1036). In practice, the state has an 18 cents per gallon tax on gasoline and petroleum products, 1 cent of which is designated for this fund (Christman 1995). This tax revenue is split between the SAF and the grant account: nine-tenths of net revenues are credited to the SAF and one-tenth is credited to the grant account, which is capped at $5,600,000. When this limit is reached, all funds are credited to the SAF. The SAF program terminates December 31, 2003 (A.R.S. § 49-1056); the 1 cent tax will continue to be levied but will be directed to the state's general fund.

There were 3,118 sites with LUSTs in Arizona, as of December 31, 1995, requiring remediation: 423 or 13.6 percent were in Pima County, and 1,688 or 54.1 percent were in Maricopa County. It is estimated that cleanup costs for these sites could reach $467 million. National figures show that one in four USTs are leaking. Tara Roesler, the current Arizona underground storage tank section program manager, projects that an additional 2,007 of the 24,000 tanks in Arizona are leaking but unreported, which would require an additional $304 million to clean up. Roesler estimates that it will take 24.5 years to generate the funds necessary for cleanup using the 1 cent tax; however, the program ends in 2003. According to Roesler, 20 percent of the SAF's annual revenue covers personnel and claims administration, with the remainder being spent for cleanup (Kohout 1995b).

Claims made to the SAF and funds paid out, inception to date, December 31, 1995. The SAF program began processing applications September 22, 1992.

Application Data		
Applications received[1]		2,241.00
No. of sites with leaking UST on which applications received[1]		1,130.00
Total amount requested	$120,290752.37	
Reductions against applications[2]	$33,653,965.10	
Net amount requested		$86,636,787.27
Estimated additional work at these sites[3]		$115,149,089.95
Projected net amount requested at these sites		$201,785,877.22
Maximum amount fund is liable for on these applications		$198,980,000.00
Average application amount		$53,677.26
Payment Data		
Application amount paid to date		$46,363,714.65
Number of applications processed to date[1]		1,681.00
Number of sites with leaking USTs on which payments made[1]		877.00
Unencumbered cash balance		$33,035,367.16

Source: ADEQ report filed with EPA administrator, L. Glascoe.
[1]On one site, remediation can be done in stages with an application filed for SAF reimbursement at each stage; one site can have numerous applications filed for reimbursement.
[2]Not all costs are reimbursable by the SAF.
[3]ADEQ estimates how much more money may be requested on pending applications.

Changes were made in the laws governing the SAF during Arizona's 1995 Special Legislative Session to protect the fund. Prior to this legislation, owner/operators who submitted applications could request coverage up to $130,000 with a $5,000 deductible (or $145,000 for costs incurred in 1991 and before) or coverage up to $1 million with a $25,000 deductible (A.R.S. §49-1054, prior to 1995 amendments). The concern was that owner/operators did not have the correct incentives to hold cleanup costs to a minimum, since no matter what the total cost, up to the limit of the coverage, they would necessarily have to pay the deductible. The new law replaces the deductible with a 10 percent co-payment required from the applicant; the SAF

will reimburse 90 percent of approved costs up to a limit of $250,000 for costs incurred before Jan. 1, 1992; $225,000 for costs incurred between January 1, 1992 and July 1, 1996; and $1 million for all claims after that time (A.R.S. §49-1054, amended HB 2001, Forty-second Legislature, Fourth Special Session 1995).

Not all costs of remediation will be reimbursed from the SAF. All applications are reviewed and only "reasonable and necessary" costs are considered (A.R.S. §49-1054). Another concern is that previously, owner/operators would have work carried out on their sites and, upon submitting applications to the SAF, find that not all costs were reimbursable. In order to protect owner/operators from incurring unnecessary costs, ADEQ will now require preapproval plans to be submitted and will inform applicants, based on these plans, which costs are reimbursable and which are not (Les Glascoe, personal communication 1996; Dulaney 1996).

The EPA must approve the use of SAF funds as a means for partially meeting financial responsibility requirements. At present, the EPA has approved allowing owner/operators to declare eligibility for $225,000 of SAF coverage as part of establishing financial responsibility. Many owner/operators have been able to obtain some form of private insurance to complete the total required coverage, but the insurance has had a $225,000 deductible. The allowable coverage may be increased to $1 million if EPA approval is granted (Les Glascoe, personal communication 1996). Subsequent to the imposition of the original financial responsibility regulations, small independent station owners found themselves unable to obtain liability insurance to satisfy the financial responsibility requirements, as insurance companies have shown great reluctance to enter this market. In addition, in the event of contamination, owner/operators who have not met financial responsibility requirements will not be eligible for SAF money.

Conclusion

There is no reason to expect the demand for petroleum storage to decline. The historical inability to prevent leakage from USTs, combined with their ubiquity, represents a significant threat to groundwater. In Pima County, significant impacts are from petroleum contamination in soil and shallow groundwater beneath the downtown Tucson area, gasoline from 23 leaking 12,000-gallon tanks at the city of Tucson's Price service center which threaten south-side wells (Bagwell 1993), a plume (area of contamination) of diesel fuel discovered in the regional aquifer beneath the center, and one city drinking water production well, Well B-78A, closed due to petroleum contamination. However, 81 percent of the USTs currently operating in Pima County were installed prior to 1988 (when technical regulations became effective) and may be expected to leak. The deadline for upgrade/replacement is 1998.

For most UST sites, since leaks generally result in subsurface contamination, exposure would generally result from inhaling vapor emissions from subsurface soils. Petroleum constituents can also leach through soils into groundwater. The EPA has typically downplayed the long-term cancer risk of petroleum constituents in groundwater because, while in the short run human exposure can cause nausea and dizziness, typically people can smell gasoline in water and will not consume it over a long period of time. However, contaminated drinking water increases costs for consumers who must find alternative drinking supplies and increases the costs of testing and treatment as mandated by the SDWA.

The statutes governing the use of underground storage tanks address the costs of increasing expected health damages due to contamination and the increased costs of supplying safe drinking water by imposing these costs on owner/operators of USTs through technical, monitoring and maintenance requirements (to reduce contamination incidents), and through requiring soil and groundwater cleanup to prevent drinking water contamination. Funds for cleanup come from private parties as well as from state funds. It is important to note that the state funds used are generated by a tax on petroleum products so that consumers as well as suppliers share in the cost of protecting health and the environment.

The success of preventative measures depends very much on how quickly existing tanks can be upgraded, how safe technology can make new tanks, how effectively monitoring for leaks and spills is carried out, and how safely the tanks are used. At present, owner/operators who do not comply with preventative measures are excluded from applying for SAF money in the case of accidents; however, because of the limited staff of the Inspections and Compliance Unit, inspection cannot be enforced.

Once contamination occurs, the level of cleanup required by statute has been costly to achieve. The financial burden on private parties and the drain

on state funds has led to increasing flexibility in determining how clean a site should be in order to be protective of health and the environment. To this end, when determining how much remediation is necessary, either predetermined standards may be used, which are based on the future use of a site and the potential threat to groundwater posed by contamination, or a site-specific risk assessment may be carried out to establish the level of cleanup, which may be less costly to achieve than complying with predetermined standards.

Comparing the costs of remediation efforts, even with more flexible means of determination, to the costs of prevention has led many regulators to express the belief that a greater percentage of limited manpower and funds should be spent on reduction and prevention of contamination (upgrading, monitoring, and inspection of facilities) and a smaller percentage on remediation of contamination.

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TRAVERTINE GEOMORPHOLOGY OF FOSSIL CREEK

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Fossil Creek is a tributary of the Verde River immediately south of the West Clear Creek drainage that flows southwest from the Mogollon Rim. Bordering on the Tonto and Coconino national forests, it ranges in elevation from approximately 2,212 m to 777 m at the Verde River. A baseflow in excess of 1.22 m^3/sec, derived mostly from a large, calcium carbonate-rich spring system, made Fossil Creek, its riparian ecosystem and geomorphic system, one of the most unique of the Verde's tributaries (Mathews et al. 1995).

The geomorphology and riparian system of Fossil Creek were unique due to the extensive system of travertine deposits below Fossil Springs. Travertine, which is identical to the mineral calcite, is calcium carbonate deposited from supersaturated water. Deposits are rare in Arizona, and areas with travertine such as Havasu Creek and Tonto Natural Bridge are valued natural wonders.

The naturalist Charles Lummis reported large travertine deposits in Fossil Creek in the late 1800s (Lummis 1891). He stated that "like hundreds of other springs in the West, they are so impregnated with mineral that they are constantly building great round basins for themselves, and for a long distance flow over bowl after bowl." Lummis did not comment on the size of the travertine "bowls," but a later visitor, F.W. Chamberlain (1904) did. He noted that "the largest pools seen were 50 to 60 yards long, 20 to 30 feet wide, and approximately 20 feet or more deep."

Construction of hydroelectric power plants at Irving and Childs in the early 1900s required diversion of this baseflow into a system of flumes, syphons, and penstocks needed to drive the electricity-generating turbines at both power plants (Mathews et al. 1995). The result was a dramatic dewatering of Fossil Creek, and degradation of the travertine geomorphic system that had created the rich riparian ecosystem.

In 1994 the Federal Energy Regulatory Commission began a review of Arizona Public Service Company's license to divert flow from Fossil Creek for hydroelectric power generation. Both the Coconino and Tonto national forests have requested restoration of all or part of the streamflow below Fossil Springs. The purpose of this paper is to examine the travertine geomorphology of Fossil Creek below Fossil Springs prior to the diversion of flows, and to assess the rate of travertine rebuilding upon restoration of springflow to the main channel of Fossil Creek.

Geological Setting

Fossil Creek is located at the southern end of the Mormon segment of the Mogollon Rim (Pierce 1987). It is characterized by Miocene-aged volcanic material (9.3 to 10.2 million years) overlying relatively level Paleozoic (Mississippian, Pennsylvanian, and Permian) sedimentary rocks of the ancestral rim. To the south of Fossil Creek, the Tonto segment of the Mogollon Rim is relatively free of volcanic material. This whole sequence is best viewed on the north wall of Fossil Creek Canyon.

The stratigraphic nomenclature for the Paleozoic sequence seen in Fossil Creek Canyon was recently revised by Blakey (1990). The Kaibab Limestone formation is mostly eroded away, leaving the Coconino Sandstone, the Schnebly Hill Formation, the Supai Formation, and the Naco Formation overlying the Mississippian Redwall Limestone.

The Naco Formation consists of Pennsylvanian-aged intercalated limestone, limey mudstone, and some sandstones with numerous unconformities. At Fossil Creek, the formation is near its northern edge, with approximately 10 percent carbonate rocks, and a thickness of about 100 m. It thickens to 200 m as it tilts to the southeast in the Fort Apache area (Blakey 1990).

On the southern edge of the Colorado Plateau, faulting and plateau uplifts are important processes in the cutting of canyons. At Fossil Creek, only the latter appears to be important (Pierce 1987). Several uplift episodes, most notably the

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late Miocene Colorado Plateau uplift, contributed to the cutting of Fossil Creek down to the level of the Naco Formation.

Fossil Springs

At about the 1280 m elevation, a series of springs that are supersaturated with carbon dioxide flow out of the Naco Formation over a distance of 330 m. Spring-generated baseflow below the springs is fairly constant at 1.22 m^3/sec. The springflow has a temperature of 22 degrees Celsius throughout the year.

Fossil Springs generates about 72 percent of the total volume of water yielded by the Fossil Creek watershed above the Irving power plant (Loomis 1994). Flows exceed the 1.22 m^3/sec spring baseflow about 23 percent of the time, mainly during snowmelt runoff and summer monsoon thunderstorms. Peak flows range from 14.47 m^3/sec for storms with return intervals of 2 years to 402.88 m^3/sec for 100-year storms.

Travertine Geochemistry

The term travertine refers to calcium carbonate materials deposited by spring waters and streamflow on in-stream plant materials (algal mats, aquatic plants, etc.), woody debris, or alluvial debris (Julia 1983). Travertine is defined as a "hard, dense, finely crystalline, compact or massive but often concretionary limestone, of white, tan, or cream color, often having a fibrous or concentric structure and splintery fracture, formed by rapid chemical precipitation of calcium carbonate from solution in surface and groundwaters, as by agitation" (Gary et al. 1972).

Travertine typically forms at, or downstream from, locations where carbonate groundwater emerges as springs (Hoffer-French and Herman 1990). The partial pressure of CO_2 in carbonate-saturated groundwater is usually 10 to 100 times greater than atmospheric levels, so CO_2 is outgassed by the carbonate waters in order to equilibrate with the atmosphere. Stream nick points, waterfalls, and steep gradient reaches enhance the CO_2 outgassing process.

The precipitation of travertine occurs in a two-step process. First, CO_2 is outgassed from a calcium carbonate-rich solution:

$$H^+ + HCO_3^- \text{<-----> } H_2O + CO_2 \text{ (gas)}.$$

This process results in a pH increase that triggers calcite supersaturation:

$$Ca^{2+} + HCO_3^- \text{<-----> } CaCO_3 \text{ (solution)} + H_+.$$

Eventually calcium carbonate precipitates as encrustations and travertine dams at or below turbulent sections.

Outgassing of carbon dioxide in turbulent portions of Fossil Creek resulted in precipitation of calcite, thereby creating a complex series of travertine terraces similar to or greater than those on Havasu Creek and other locations in the Grand Canyon. The reports by Lummis (1891) and Chamberlain (1904) certainly confirm this.

Travertine Geomorphology

Diversion of most of the baseflow to a power plant constructed at Irving in 1915 halted aggradation of the Fossil Creek channel through travertine formation. Without the input of calcium carbonate-rich baseflows, the travertine terrace system began to deteriorate. Subsequent stormflows and flash floods degraded the Fossil Creek channel back to its bedrock controls. Peakflows in 1915 were quite likely much larger. Photographs taken during construction of the Irving power plant indicate that the Fossil Creek watershed had far less woody vegetation than it does now. Thus, the hydrologic response of Fossil Creek would have been much greater, and the erosive power of those flows would have been considerably larger.

Preliminary reconnaissance of the 6 km reach of Fossil Creek from the diversion dam to the Irving power plant (Figure 1) indicates that there were a minimum of 81 distinct sets of travertine dams located mainly at or near channel nick points where water turbulence increases. The remaining buttresses of these dams are evident throughout that 6 km reach. Visual observations confirm that some of these terraces resulted in channel aggradations of up to 10 m above current bedrock-controlled channel bottoms.

In some instances, multiple terraces developed in the riparian zone along Fossil Creek (Figure 2). Excavations indicate that these terrace systems were controlled by travertine dam and pool formation. Travertine pools were filled with alluvium during episodic peakflows, allowing the channels to aggrade and terraces to build at increasingly higher levels.

Although peakflows and flash floods are a major degradation process in the travertine geomorphology, the rate and occurrence of substantial degradation is uncertain. When travertine formation was terminated by diversion of the spring flows, these high-velocity events certainly accelerated degradation of the Fossil Creek channel. In the presence of active travertine deposition, the degree of channel degradation during peakflows

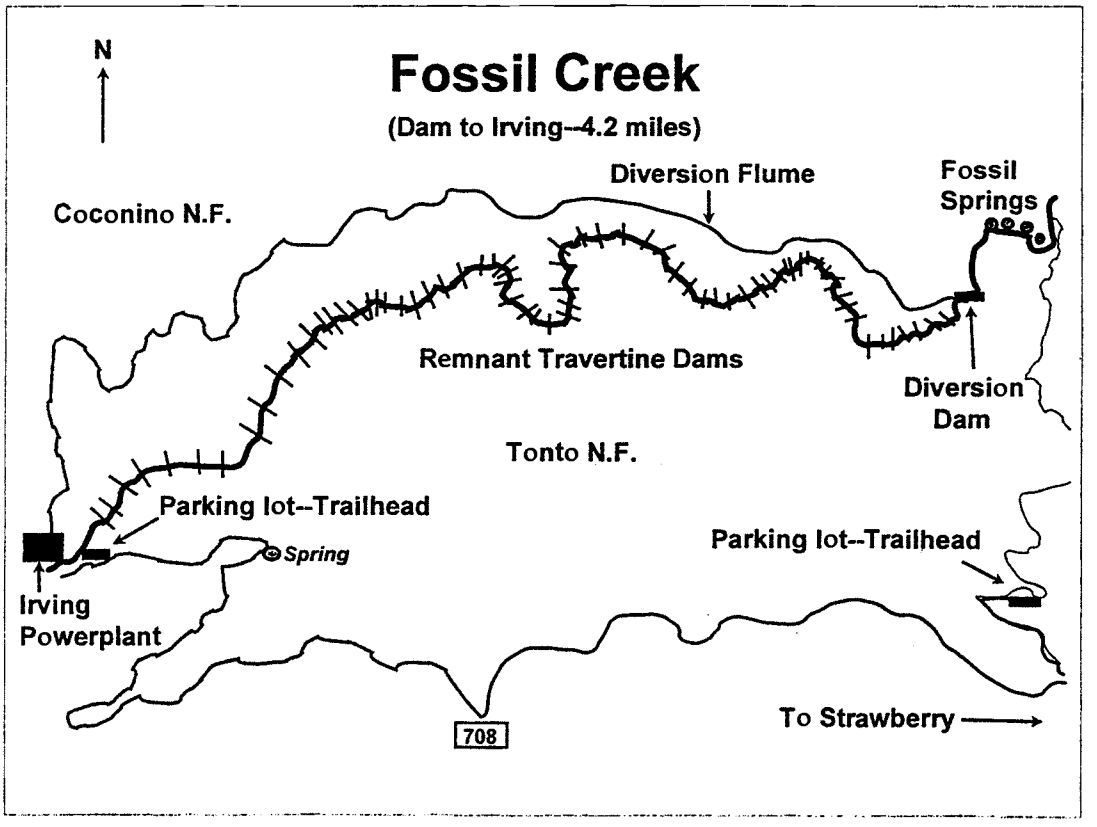

Figure 1. Sets of travertine dam remnants mapped by GPS in Fossil Creek between the Irving plant and the diversion dam.

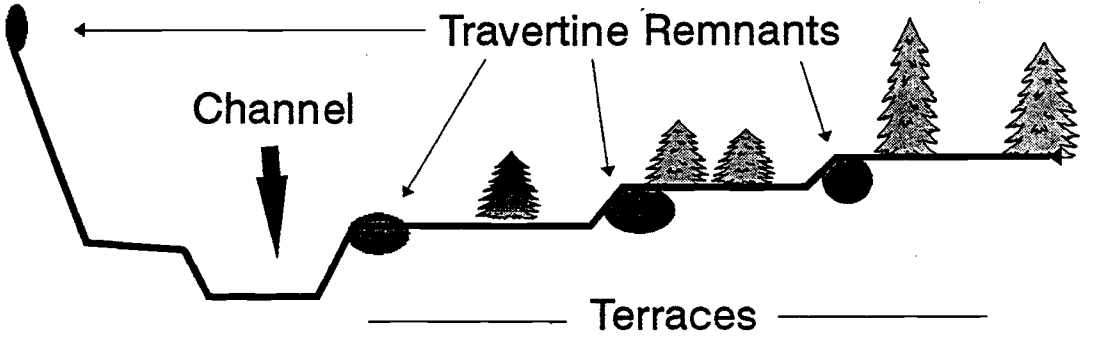

Figure 2. Cross-section showing the relative locations of a terrace system and travertine buttress remnants.

was probably limited. Evidence of this is presented by the survival of several small travertine dam structures below Irving following high stormflows during the winter of 1995. The pools of these travertine dams filled in, but the aggradation process continued.

Rates of travertine formation in Fossil Creek have not yet been accurately determined. However, based on travertine chemistry, there is evidence of a wide range of rates rather than a uniform rate. The Draft Environmental Assessment for Hydropower License on Fossil Creek (Federal Energy Regulatory Commission and USDA Forest Service 1996) states a rate of 7 mm/yr (0.29 in) based on an analysis done by CH2M Hill consultants. On some travertine buttresses, annual rings close to 7 mm in

diameter are evident. These were probably in slacker waters and reflect a fairly constant deposition of calcite on algal mats (hence the annular nature of the rings). However, in faster waters travertine has been observed to accumulate at rates of up to 30 cm/month.

During March, 1996, the full flow of Fossil Spring was diverted back into the original channel during maintenance at the Irving power plant. During that one-month period, travertine was observed to accumulate to depths of 30 cm by incorporating woody debris flushed down the channel by resumption of normal flows. Thus it is possible that new travertine dams could form at rates far in excess of those originally believed possible and documented in the draft environmental assessment. The whole character of the riparian zone geomorphology and biotic communities dependent on water in the Fossil Creek channel could change quite rapidly.

The Future

This paper examined the geomorphology of remnant travertine dams of Fossil Creek relative to the current geomorphic setting. The effects of rewatering Fossil Creek at some time in the future could be quite dramatic in a very short period of time depending on how much water is diverted back into the original channel. The amount and timing of this rewatering is the focal point of the environmental assessment currently being conducted (Federal Energy Regulatory Commission and USDA Forest Service 1996). Physical evidence from the March, 1996, rewatering of Fossil Creek indicated that accretion of travertine and building of travertine dams in the presence of woody debris can be quite rapid (30 cm/month). Predictions of travertine dam formation based purely on calcite chemistry greatly underestimate the rate of travertine dam buildup. Restoration of the Fossil Creek riparian and travertine dam system would most likely occur very rapidly when flows from Fossil Springs are returned to the original channel.

Several studies are in progress or planned for the future to improve our understanding of the role of travertine formations in the Fossil Creek riparian ecosystem. Evidence from the past indicates that a large, diverse riparian ecosystem flourished in Fossil Creek before watershed disturbances and diversion of streamflow took their toll. The first step in restoring the Fossil Creek riparian ecosystem will be the return of carbonate-saturated springflow. Recovery is likely to be rapid, and Arizonans will again be able to see one of this state's truly natural wonders.

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RELATIONSHIPS OF NATIVE FISHES AND AQUATIC MACROHABITATS IN THE VERDE RIVER, ARIZONA

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

There is limited information on the relative, quantitative and physical habitat utilization of native fishes within large desert river systems in the Southwest. Rinne (1992) delimited habitat for a native fish community in a small desert stream, Aravaipa Creek, and presented limited information on two native species in larger southwestern rivers (Rinne 1989, 1991). However, studies of the fish fauna in a large river are currently in progress on the upper Verde River, Yavapai County, Arizona (Stefferud and Rinne 1995). These studies, in part, are designed to define habitat affinities of native fishes in this reach of yet undammed, free-flowing river. Six native species remain in this reach, plus an equal number of non-native, introduced species (Stefferud and Rinne 1995). All appear to occupy habitats defined largely by velocity, and to a lesser extent, depth of water. Because of the paucity of information, velocity and depths of five qualitatively defined aquatic macrohabitats are being collected in the upper Verde River. In part, this ongoing study is designed to define aquatic macrohabitats in this reach of river and to delimit fish utilization of these habitats.

This paper has three objectives. The physical characteristics of these aquatic macrohabitats are defined, the associations with and utilization of these habitats by the respective species are delimited, and the significance of these data and their management implications are evaluated.

Study Area

An introduction to and a map of the general study area can be found in Stefferud and Rinne (1995). In brief, the upper Verde River is one of few remaining undammed rivers in the Southwest. Base flows range from 0.57 m^3/sec at the Paulden stream gage in the upper portions of the study areas to 1.42–1.70 m^3/sec at the Sycamore Creek site (Stefferud and Rinne 1995; Rinne and Stefferud in press). The study of this stream reach is programmed for 10 years.

Methods

Data have been collected at seven established sampling sites since spring 1994, following a 50± year flood event in winter of 1992–93. The data presented represent the first two years (1994–95) of a 10-year programmed effort. Study reaches are sampled twice a year, in spring (April) and autumn (October). All seven sites are sampled in the spring; however, to reduce sampling stress on fish populations, only the uppermost and lowermost sites are sampled during autumn. Aquatic macrohabitats are qualitatively defined and physical measurements are normally taken in spring only. However, in the event of summer flooding, which may alter physical habitat, physical measurements would also be taken in autumn. We have visually (qualitatively) designated four major lotic or flowing water habitats: high-gradient riffles, low-gradient riffles, runs, and glides. Almost a dozen different pool habitats are being used in describing lentic or still-water macrohabitats in this paper; we classify all into one category—pool. Length of each habitat is measured and three equidistantly spaced transects recording width, depth, and velocity are measured each spring. Initially, only general categorization of stream substrates was made.

In spring 1996, we commenced to move from qualitatively to quantitatively defining macrohabitats in the Verde River. We sampled 14 microhabitats (see Neary et al. this volume) at the Burnt Ranch site in order to more validly and unequivocally define macrohabitats (see Figures 1 and 2). Accordingly, laser survey technology was employed to more accurately and quantitatively define gradients of the respective habitat types. In addition, pebble counts (Bevenger and King 1995) are being used to better define substrate composition of each habitat. Four major substrate cate-

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gories are employed in analyzing substrate: sand (≤2 mm), gravel (3–64 mm), pebble (65–126 mm) and cobble/boulder (>126 mm). Fishes in each habitat were captured using a combination of backpack DC electrofishing gear and seines (3–13 mm mesh). Only fishes captured in 1994–95 sampling were used to define macrohabitat use by respective species (see Figures 3–5).

Results

Based on visual (qualitative) designation, pools and glides had the greatest mean depth and the least mean velocity (Table 1). The range of velocity and depth of the four lotic macrohabitats varied widely; glides were greatest in mean and median depth, and low-gradient riffles were least (LGR). No trend was obvious for depth of these habitats. Mean velocity of lotic habitats increased from glides through high-gradient riffles (HGR).

By comparison, laser-estimated gradients at the 14 microhabitat sites parallel mean velocities of the habitat types (Figure 1). A working hypothesis for categorizing macrohabitats based on percent gradient is glides = ≤0.3%, runs = 0.3–0.5%, LGR = 0.6–1%, and HGR >1%. Mean velocities also fall within these same categories. Substrate composition estimated within the 14 microhabitat sites is directly legislated by velocity of water (Hynes 1972). Substrate also demonstrated patterns relative to the major habitat types in the Verde. That is, sand and gravel decreased from glides through HGR habitats and pebble and cobble/boulder substrate increased in the same sequence (Figure 2).

Table 1. Mean velocity and depth characteristics of aquatic macrohabitat types in the upper Verde River, 1994. Number of macrohabitats sampled is in brackets, and medians are in parentheses.

Habitat type	Depth (cm)	Velocity (cm/sec)
Pool [39]	46 (40) 20–90	22 (2) 0–70
Glide [60]	55 (50) 25–43	22 (25) 15–26
Run [94]	42 (37) 20–80	48 (43) 20–80
Low-Gradient Riffle [96]	25 (25) 10–41	59 (51) 30–80
High-Gradient Riffle [98]	34 (30) 20–60	87 (90) 76–150

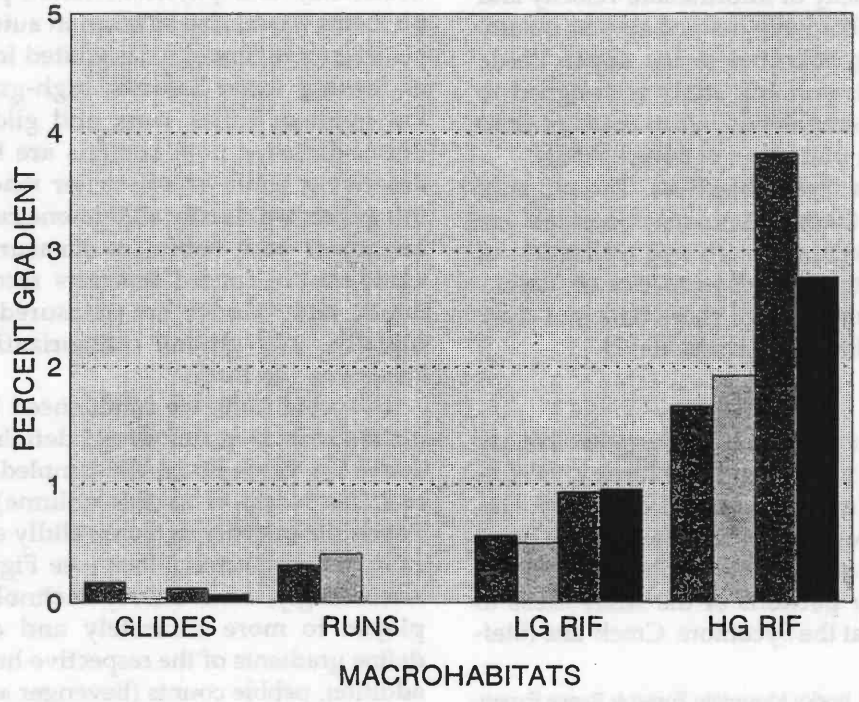
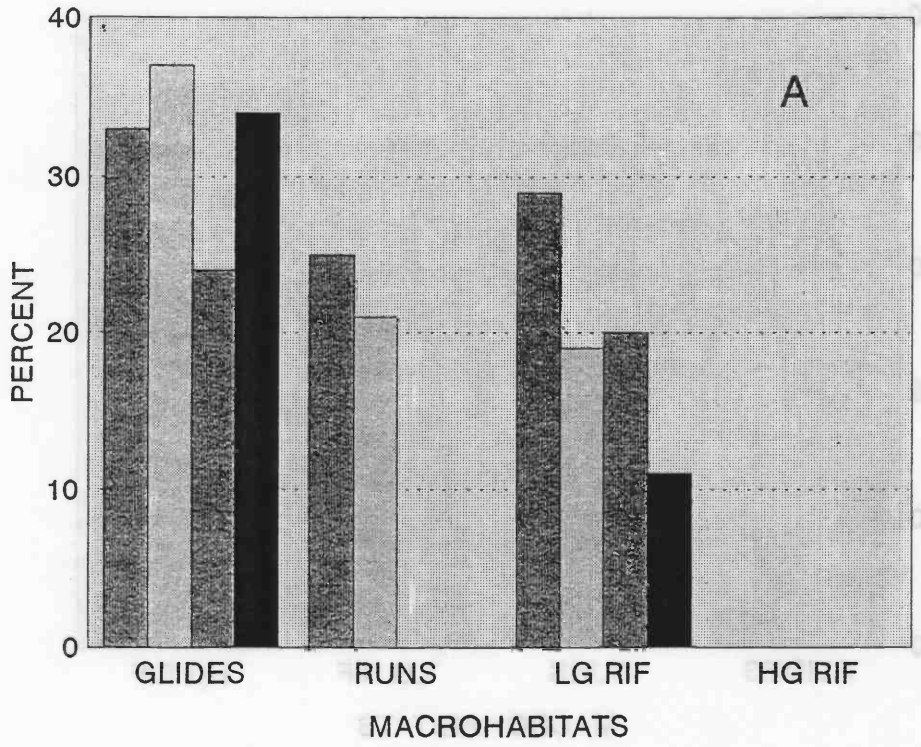


Figure 1. Comparison of laser-generated gradients at the 14 microhabitats at Burnt Ranch, upper Verde River, spring, 1996.



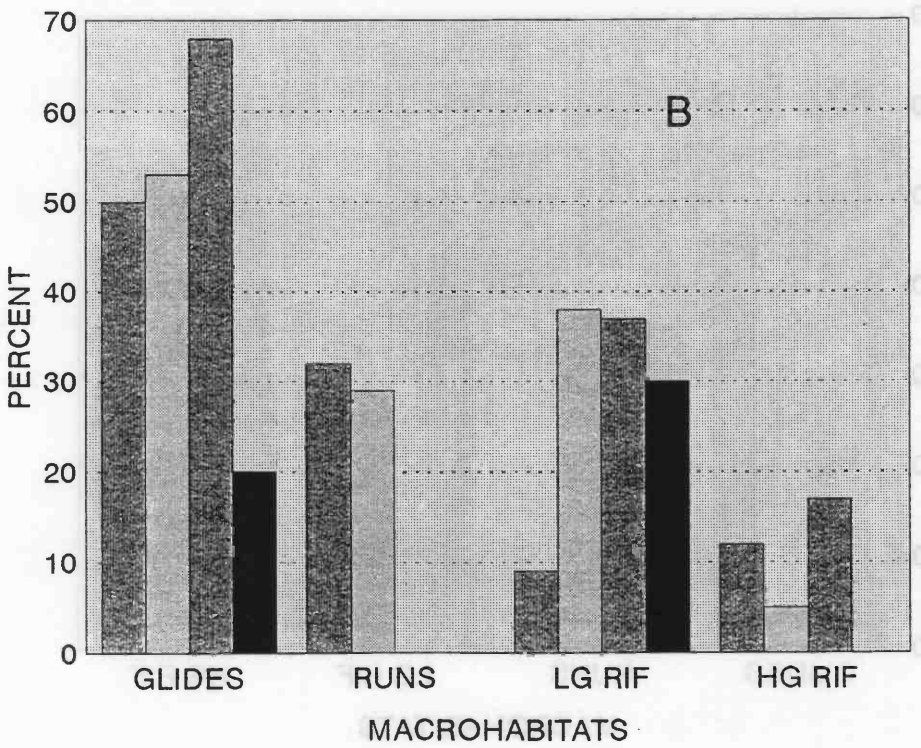
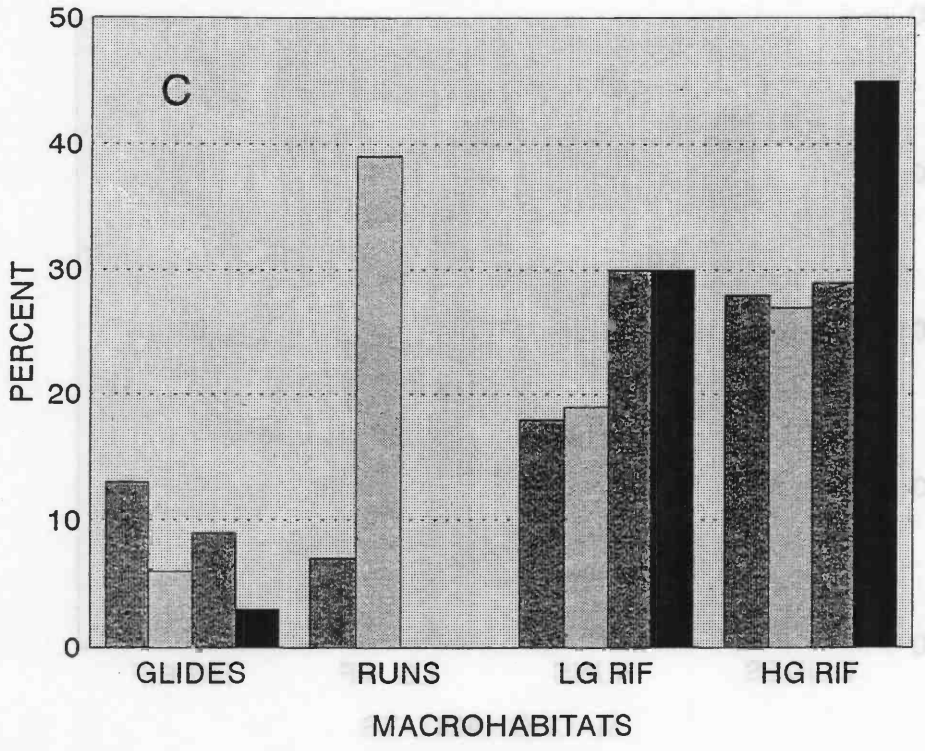
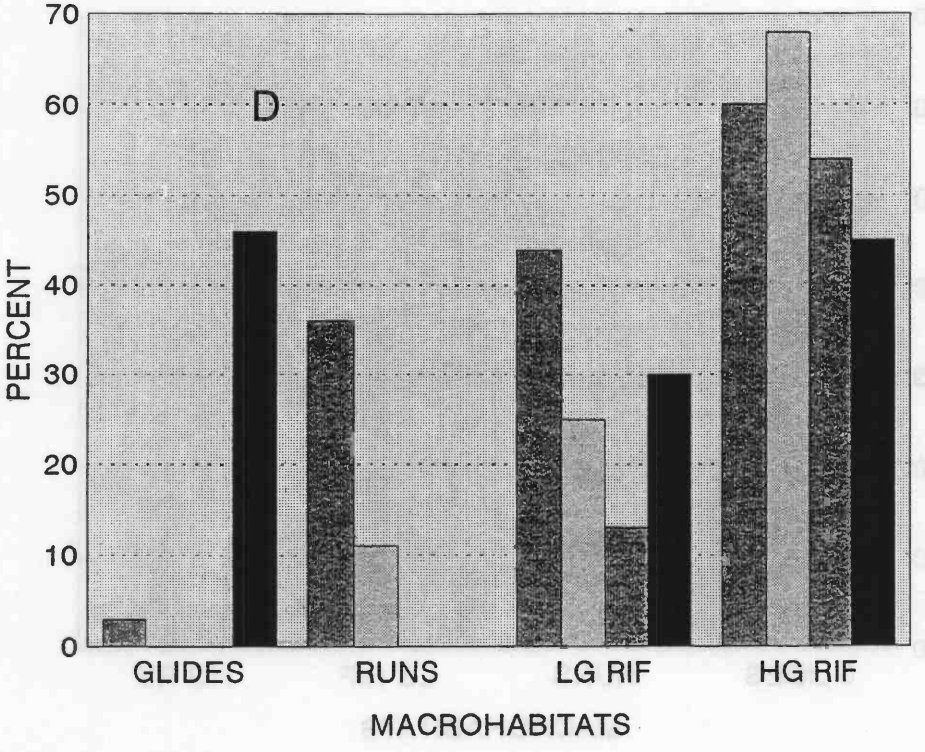


Figure 2. Comparison of (A) sand, (B) gravel, (C) pebble, and (D) cobble/boulder components in 14 aquatic microhabitats in the upper Verde River, spring 1996.





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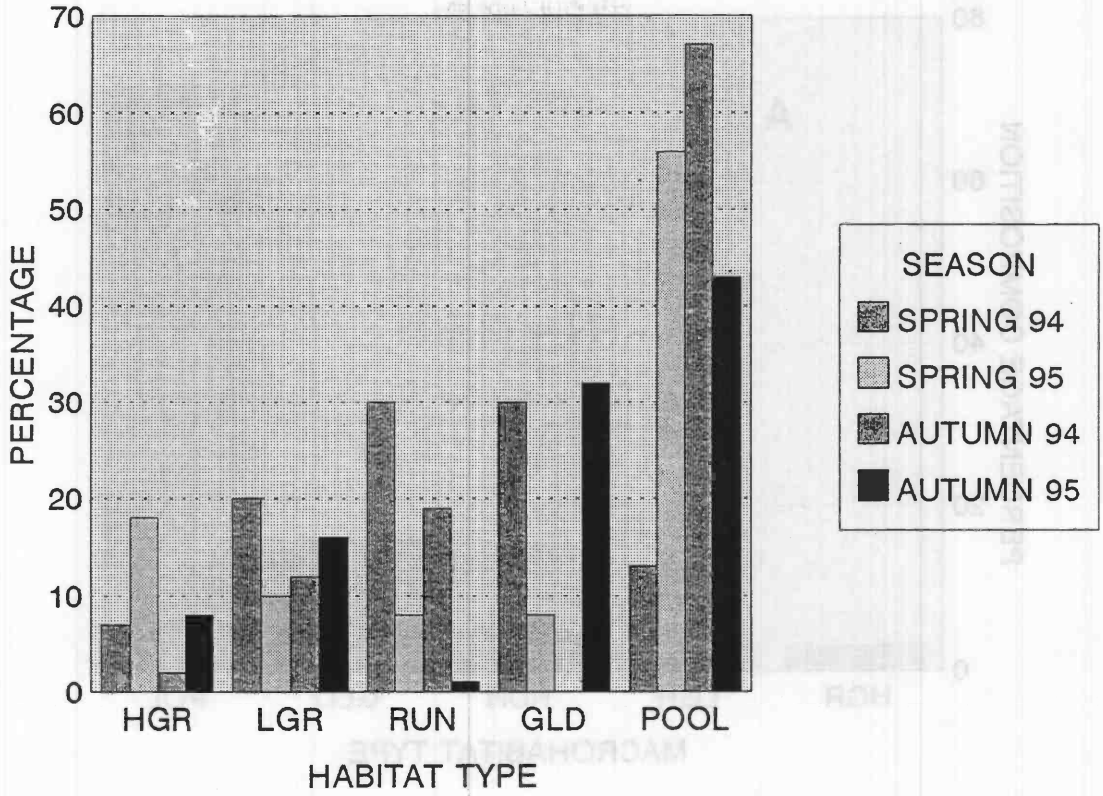


Figure 3. Comparison of macrohabitat utilization by the native fish community in the upper Verde River, 1994–95.

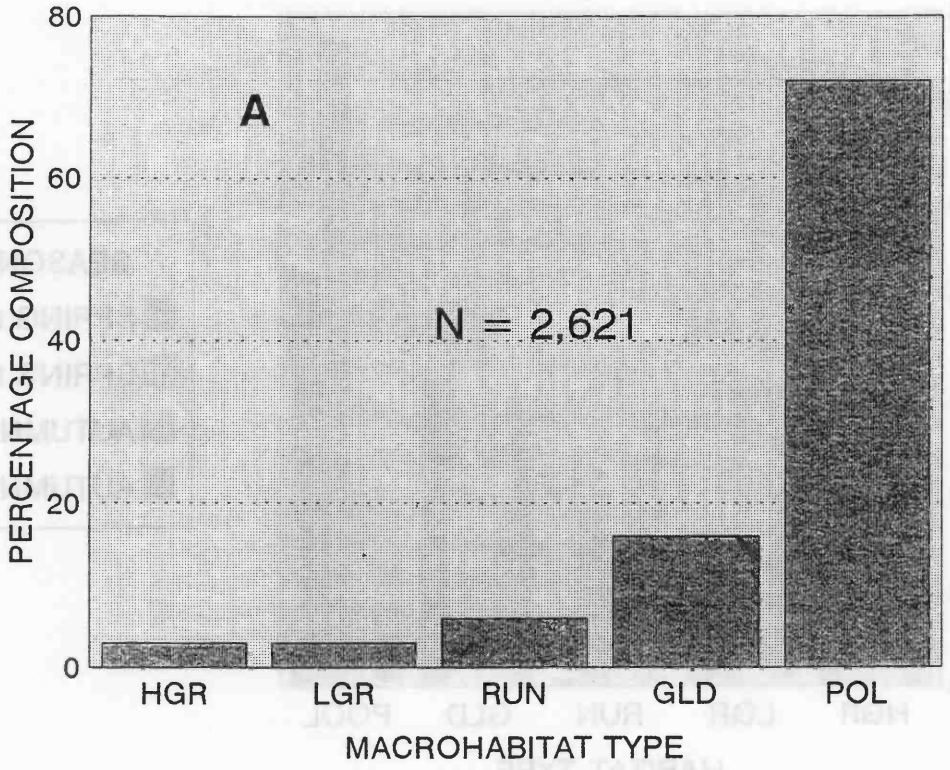
Information collected on fish numbers at the seven sites over four sample periods gives the appearance that, with the exception of spring 1994 sampling, the native fish community is a pool-dwelling assemblage (Figure 3). However, this is attributable to occupation of pool type habitats by two of the three larger sized species, Sonora sucker and roundtail chub (*Catostomus insignis* and *Gila robusta*, respectively), which comprised almost half of the total number of fish collected during the four sampling periods (Figure 4). By comparison, over half (56%) of desert suckers utilized HGR and LGR macrohabitats. Similar to desert sucker, most (90%) speckled dace also were collected in HGR and LGR habitats (Figure 5). Longfin dace predominantly occupied LGR and run habitat. Finally, most (80%) of the federal and state-threatened spikedace were collected in run and glide habitat (Figure 5).

Discussion and Conclusions

Qualitatively defined aquatic macrohabitats appear to be reasonably valid based on velocity, substrate, and gradient criteria at the 14 micro-habitat sites. However, more data are needed. For example, physical similarity of pools and glides suggests that refinement is needed for these two habitat types. Also, based on macrohabitat data collected in 1994–95, spikedace preferred runs and glide habitats. Microhabitat sites suggested that spikedace prefer runs and low-gradient riffles (Neary et al., this volume).

A working categorization for gradients and velocities is proposed; however, more quantitative microhabitat data are needed to validly and unequivocally define the respective macrohabitats. Depth appears to be much less of a definitive physical characteristic than velocity or gradient for macrohabitat designation. We acknowledge that only limited microhabitat data (spring 1996) are available on substrate characteristics. However, substrate is certainly delimited by stream gradient and velocity of water; this physical feature may be very important both in habitat definition and in legislating fish distribution.

Gradient may be especially important in defining habitats for the habitat specialists such as spikedace. The importance of substrate in spike-



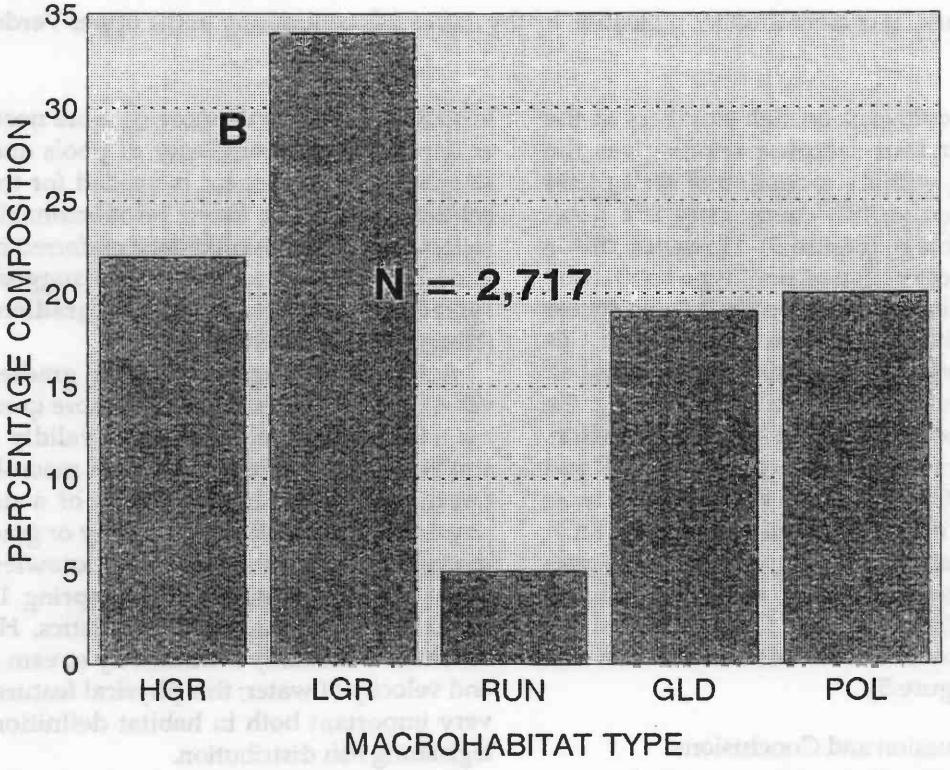
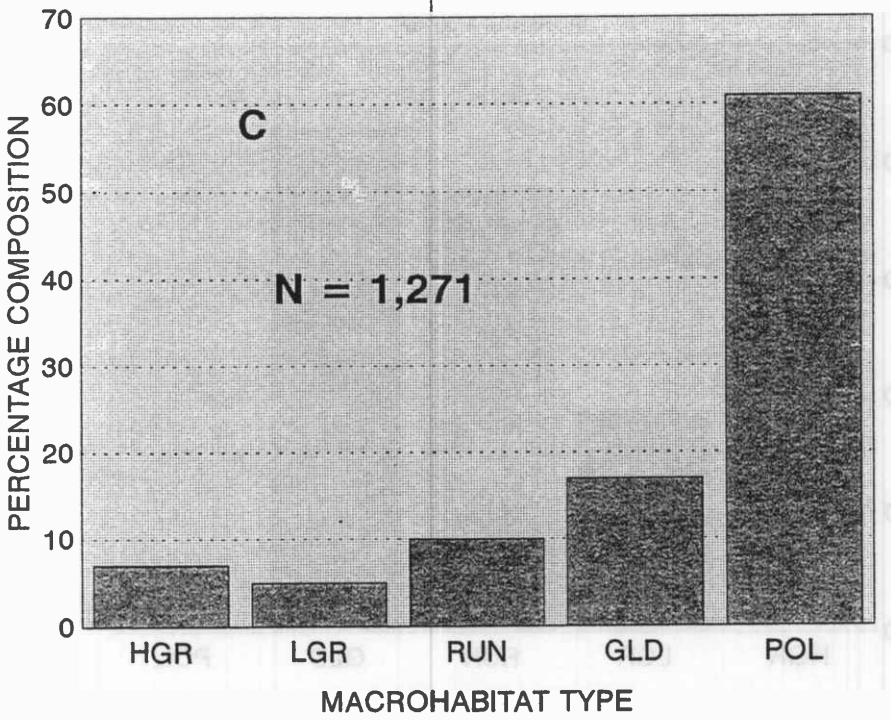


Figure 4. Macrohabitat utilization of the (A) Sonora and (B) desert suckers and (C) roundtail chub in the upper Verde River, 1994–95.



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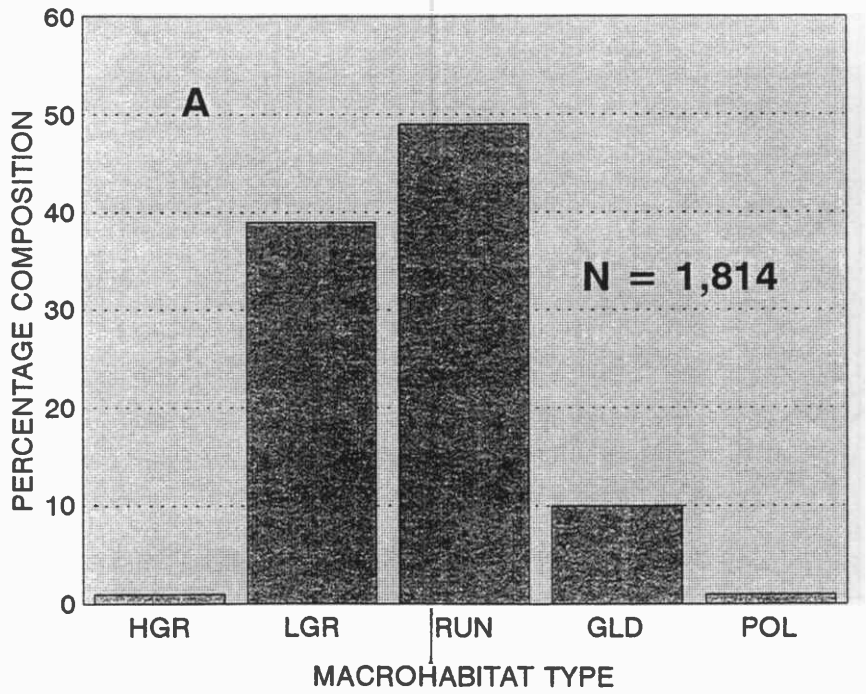
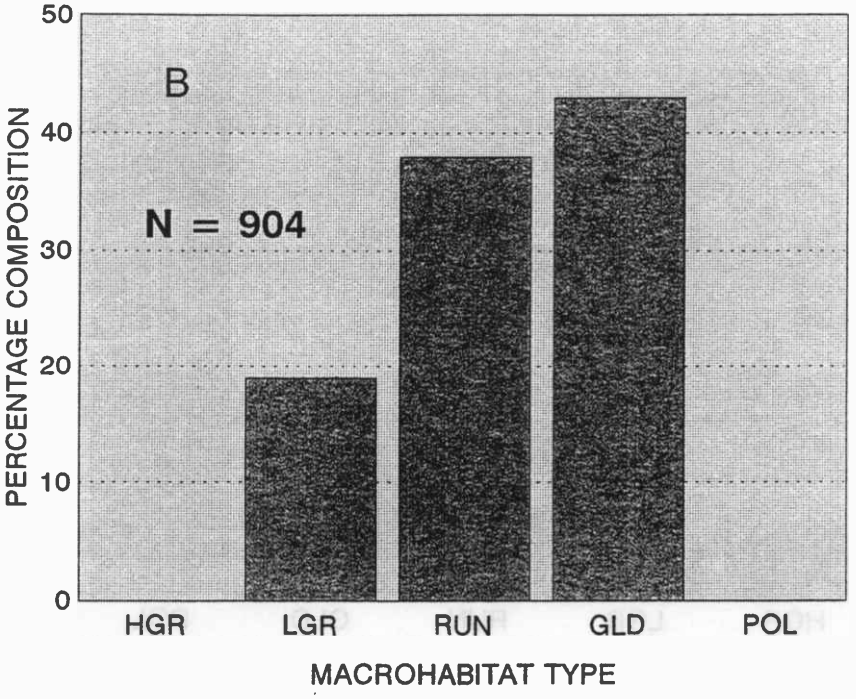


Figure 5. Macrohabitat utilization by the (A) longfin dace, (B) spikedace, and (C) speckled dace in the upper Verde River, 1994–95.



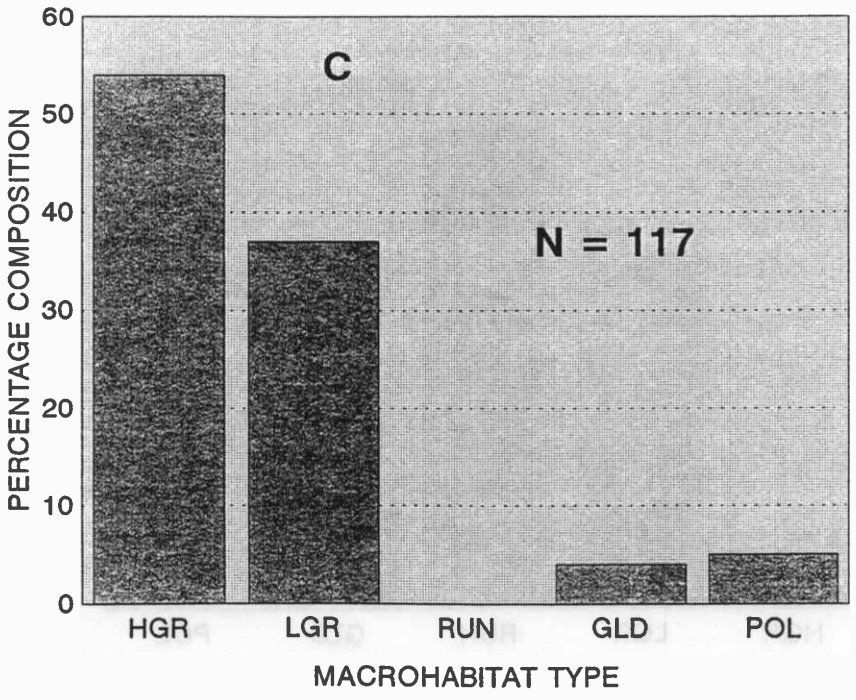


Figure 5 (concluded)

dace distribution and abundance is beginning to be documented for this species in the Verde River (Neary et al., this volume) and substantiates earlier work by Barber et al. (1966) and Barber and Minckley (1967). This threatened species occupies gravel to pebble substrates (Rinne 1991, 1992; Neary et al., this volume). Both our spring 1996 data in the upper Verde River and that collected in April 1985 (Rinne 1991) indicate that gravel/pebble (3–64 mm) substrates are particularly important for spawning habitat of spikedace.

Our data on the upper Verde River suggest that gravel/pebble substrate habitats are concentrated in runs and low-gradient riffle habitats (Figure 2). In part, gradient and the resulting increase in velocity are primary influencing factors, but channel narrowing also increases velocity. That is, if the same volume of water passes through a narrower and deeper channel, velocity is increased and the deposition of gravel/pebble substrate is enhanced, whereas the deposition of sand substrate decreases. The process of channel narrowing requires streambank stability. Herbaceous native aquatic vegetation has been suggested as a very important component contributing to streambank stability (Medina 1995). Land-use practices that remove vegetation can reduce streambank stability and the channel-narrowing capacity of a stream (Neary and Medina 1995). Studies examining the linkages of grazing, streambank stability and morphology, vegetation composition and density, fish habitat, and fish populations on the upper Verde are being initiated and will conceivably provide answers on the linkage between grazing and spikedace habitat and populations. Land managers conceivably can use this information to implement grazing strategies in time and space that sustain spikedace habitat.

Our data on the upper Verde (Figure 5) and that of other authors (Barber and Minckley 1967; Minckley 1973) document the close relationship of longfin dace with low-velocity, sandy bottom habitats. The occupation of pool-type habitats by almost 60 percent of Sonora sucker and roundtail chub also suggests the importance of the lentic component of the upper Verde ecosystem. The other three species are lotic water inhabitants. In summary, the native fish community in the Verde River basically utilized the same relative macrohabitats as recorded for the same community in Aravaipa Creek (Rinne 1992). However, more data are needed on the relative proportions of the major aquatic macrohabitat types in the upper Verde river before irrefutable statements can be made on habitat use by the respective species. That is, at present, we are assuming that the habitats we have sampled at the seven sites are representative of those available in the 60-km reach of the upper Verde.

Flood events may drastically alter extant macrohabitats, resulting in a new assemblage of habitats within established study reaches. The more drastic the changes in the habitats within these reaches, the greater the changes in the relative composition of the fish community. Such events and resultant changes provide the opportunity to validate both micro and macrohabitat vs. fish relationships. Flood events such as those in winter 1993 appear essential for creating and maintaining pools in addition to inducing increased or successful spawning and recruitment (Stefferud and Rinne 1995; Rinne and Stefferud, in press).

There is a need for additional refinement in the definition of the physical characteristics of the aquatic macrohabitats in the upper Verde; however, gradient appears to be of great importance in fish distribution. Gradient can be very precisely defined by laser technology. Further, gradient directly influences the velocity of water and substrate composition. Therefore, we suggest that this physical feature may be the controlling factor for quantitatively delimiting macrohabitats in this and perhaps other desert streams and rivers. Nevertheless, additional quantitative, objective data on macrohabitat utilization by the respective members of the native and non-native fish community will be needed to compare with and validate data collected to date.

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PHYSICAL HABITAT USE BY SPIKEDACE IN THE UPPER VERDE RIVER, ARIZONA

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The spikedace, *Meda fulgida,* is a threatened species of short-lived minnow that persists only in the mid-elevations of the upper Gila River in New Mexico, and Aravaipa Creek and the upper Verde River in Arizona (Rinne 1991). The fish is mottled on the back, silvery on the sides, and whitened ventrally (Minckley 1973). Males become golden on their dorsal and lateral surfaces during breeding. Spawning is in the spring and early summer. One-year-old females produce one batch of 100 to 300 eggs, and two-year-olds may produce two in a single spawning period. Young emerge in April and May and usually grow to lengths of 35-40 mm by November. Individuals seldom exceed four years in age and 65 mm in length. Spikedace feed on aquatic and terrestrial insects as well as the fry of other species.

The upper Verde River is one of the most important habitats for the spikedace. It is a relatively wild, free-flowing stream that has low baseflows (0.57 m^3/sec), no perennial tributaries, and periodic drought and extreme flood events (Stefferud and Rinne 1995). Although adapted to southwestern streams, spikedace populations are affected by natural variations in the Verde River hydrograph (i.e. alternating floods and drought). Habitat degradation produced by impoundment construction, dewatering of streams for irrigation, channel erosion, and sedimentation have contributed to the decline of spikedace populations. In addition, introduction of non-native fishes such as the red shiner has seriously affected the species through competition for food and habitat (Rinne 1991; Douglas et al. 1994). Finally, livestock grazing has been implicated in the decline of spikedace habitats and populations through streambank deterioration and sedimentation. However, the impact of grazing relative to non-native species predation is still unknown.

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A major consideration in studying this rare species of native fish is the determination of its preferred habitat. Past surveys may have missed the spikedace because of sampling in the wrong microhabitats. Reconnaissance surveys of the upper Verde River completed in 1993 and 1994 indicated that within stream reaches actually occupied by spikedace, the fish were concentrated in discrete microhabitats.

This preliminary study had two major objectives. First, we planned to examine and define the preferred physical habitat of spikedace relative to water velocity, stream gradient, and stream substrate composition. Second, we wished to make specific sampling recommendations for future studies.

Methods

The site of the fish sampling is in the upper Verde River, Yavapai County, about 6 km downstream of Sullivan Lake. This reach of stream is known as the Burnt Ranch section (see Stefferud and Rinne 1995). It is subject to periodic drought and extreme flood events such as the 70+ year event which occurred in early 1993. Stream types within this reach include rocky, high-gradient riffles; lower gradient riffles consisting mainly of gravel; gentle runs with few large rocks, gravel, and some sand; and wide, gentle glides with mainly sandy bottoms (Rinne and Stefferud 1996).

Fish sampling and physical habitat characterization occurred on 14 different stream microhabitats within several reaches at the Burnt Ranch site. Each distinct microhabitat was classified into one of five types: high-gradient riffle (HGR), low-gradient riffle (LGR), run (RUN), glide (GLD), and pool (POL). Only two runs were found within the sampling areas, and pools were not evaluated based on previous information on the absence of spikedace in this microhabitat. This resulted in evaluation of 14 microhabitats. Each microhabitat was sampled using a 5 x 15 m seine with 3 mm mesh. Fish were collected by dragging the seine

from up to downstream in microhabitats with pebble to sand bottoms. In microhabitats with extensive cobbles and boulders (i.e. high-gradient and low-gradient riffles), fish were collected by employing direct current, backpack electrofishing gear. In this approach, the seine was used to block the downstream end of a sample microhabitat, and the microhabitat was sampled from upstream to downstream into the net using the electrofishing unit. After either procedure, the seine was brought ashore and captured fish were immediately placed into a water-filled bucket. All the collected fish were identified by species and placed on a measuring board. Total lengths from snout to tail were recorded for use in other studies, and the fish were released back into the Verde at the site of capture.

After sampling the fish in the study area, data were collected to establish the velocities, depths, gradients, and substrates of each sampled microhabitat. Equipment and established procedures to determine these physical habitat measurements are described in Rinne and Stefferud (1996). Basically, stream velocities and depths were measured at three transects within each sample reach by a direct readout current meter attached to a 2-m rod with centimeter calibrations. When the depth was established and recorded, the bottom of the current meter was then held at 60 percent of the depth from the water surface to obtain mean velocity of the microhabitat. The velocity was recorded when the current meter readout came to a constant reading. Gradients and microhabitat dimensions were determined using a Criterion 400 laser survey instrument. Once into the survey measurements screen, the Criterion was aimed at a target with a reflector that sends back the laser beam. Each target was positioned at the upstream end of the sample site with the Criterion at the downstream end to determine gradient. It was also set up across the stream to determine width. When the Criterion laser fired, information such as distance, azimuth, percent slope, and gradient were automatically computed and presented on the data screen.

To characterize stream substrates, a modified pebble count technique developed by Bevenger and King (1995) was used. Using this method, one sample of the substrate, possibly consisting of sand, gravel, pebbles, cobbles, or boulders, was picked up and measured on the longest dimension with a ruler at the end of every three steps. Thirty measurements were taken throughout the entire sampling site in a random zig-zag pattern. A simple average of the measurements gave the average sediment size for that site.

Results and Discussion

Spikedace microhabitat use was evident in results from stream sampling conducted in 1994 and 1995 (Figure 1). However, microhabitats were visually characterized during the 1994–1995 samplings, and general measurements of depth, velocity, and substrate material were made. No attempts were made in 1994–1995 to systematically classify microhabitats based on physical measurements. Some microhabitats that were identified visually as pools were actually mid-channel pools that were part of other microhabitat types. Since the ability to separate these microhabitats in a reliable manner was uncertain, plans were developed to add measurements of gradient, velocity, depth, and substrate to further refine the definition of the microhabitats.

Physical habitat information gathered during the March, 1996, survey for spikedace was compared to visual estimates of microhabitat type taken at each site. As a result of the laser survey measurements, each habitat was clearly associated with a range of gradients (Figure 2). Glides were less than 0.25 percent, runs were between 0.25 percent and 0.50 percent, low-gradient riffles were between 0.51 percent and 1.00 percent, and high-gradient riffles were greater than 1.00 percent. This gradient separation allows for a more consistent and accurate definition of habitat type than visual determinations.

Spikedace were found in greatest abundance in gradients between 0.4 and 0.6 percent (Figure 3). Accordingly, these gradients fall within runs to low-gradient riffle microhabitat classifications, as we have suggested. These two microhabitats contained almost 60 percent of the spikedace captured in the entire upper Verde River during 1994–1995 (Rinne and Stefferud 1996). Therefore, there is good agreement between the qualitatively defined habitats and those we have defined by laser surveying.

Spikedace were found in habitats with substrates of less than 10 percent sand (Figure 4). According to the greater density of spikedace present in stream reach 3, we determined that spikedace favored habitats with lower sand content. Glides, runs, and low-gradient riffle microhabitats contain increasingly less sand substrate, respectively (Rinne and Stefferud 1996). The data show that spikedace certainly prefer gravel over sand substrate during the spring spawning season. More

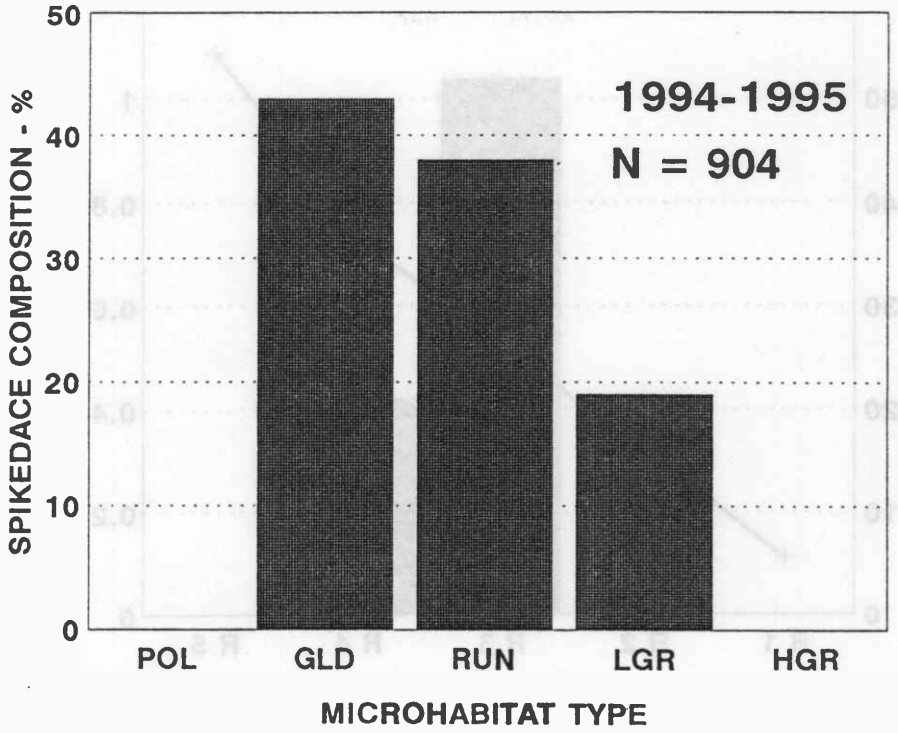

Figure 1. Spikedace distribution by microhabitat type, upper Verde River, Arizona (adapted from Rinne and Stefferud 1996).

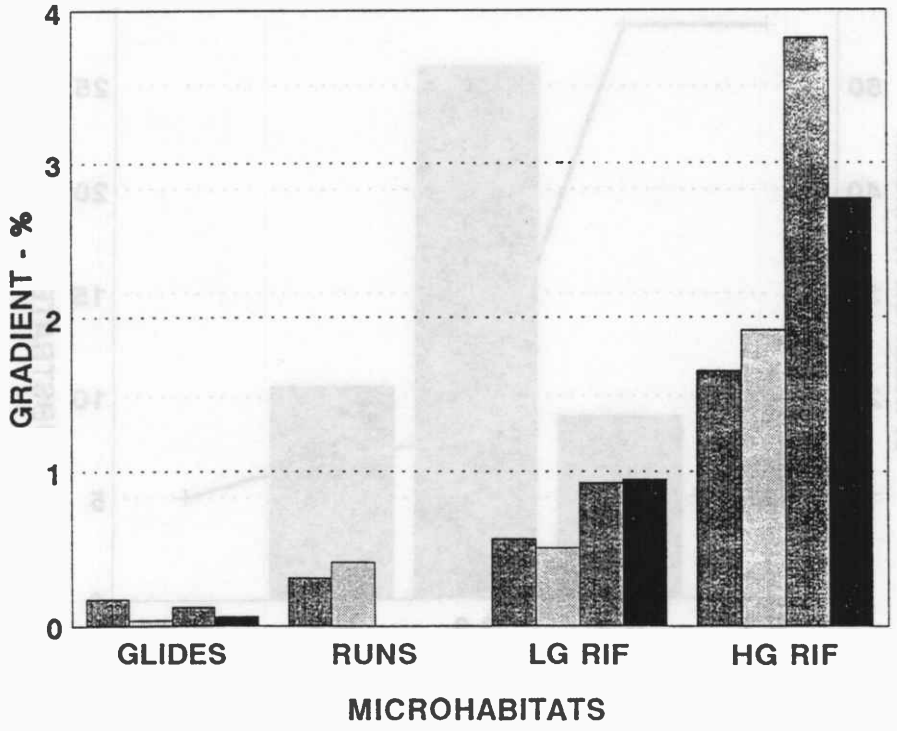

Figure 2. Comparison of gradients of 14 microhabitats, upper Verde River, March 1996.

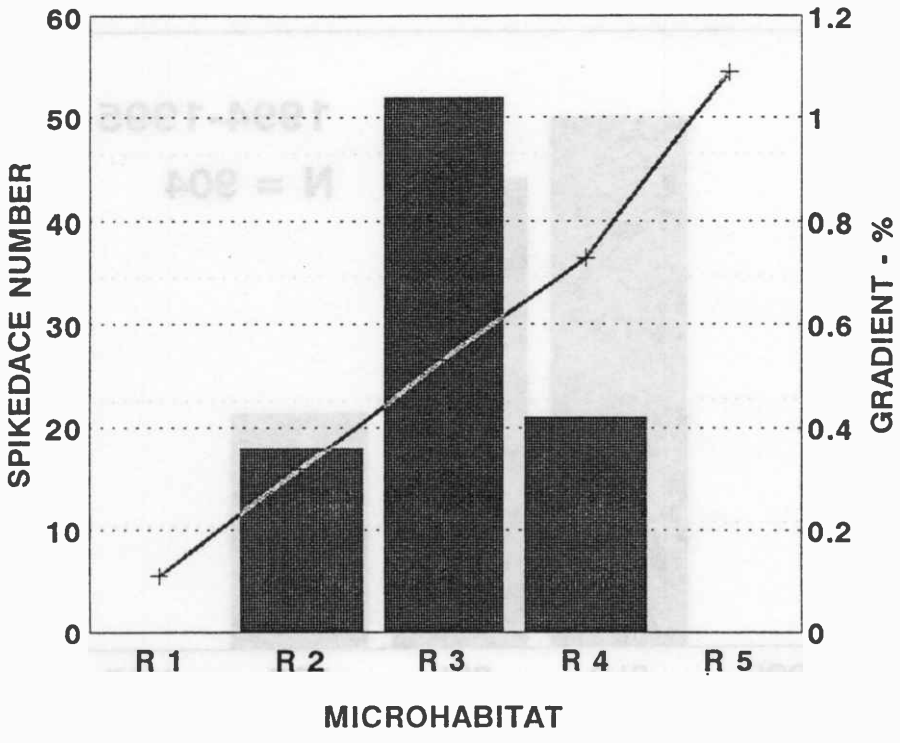

Figure 3. Spikedace numbers and stream gradient, Burnt Ranch section, upper Verde River, March 1996.

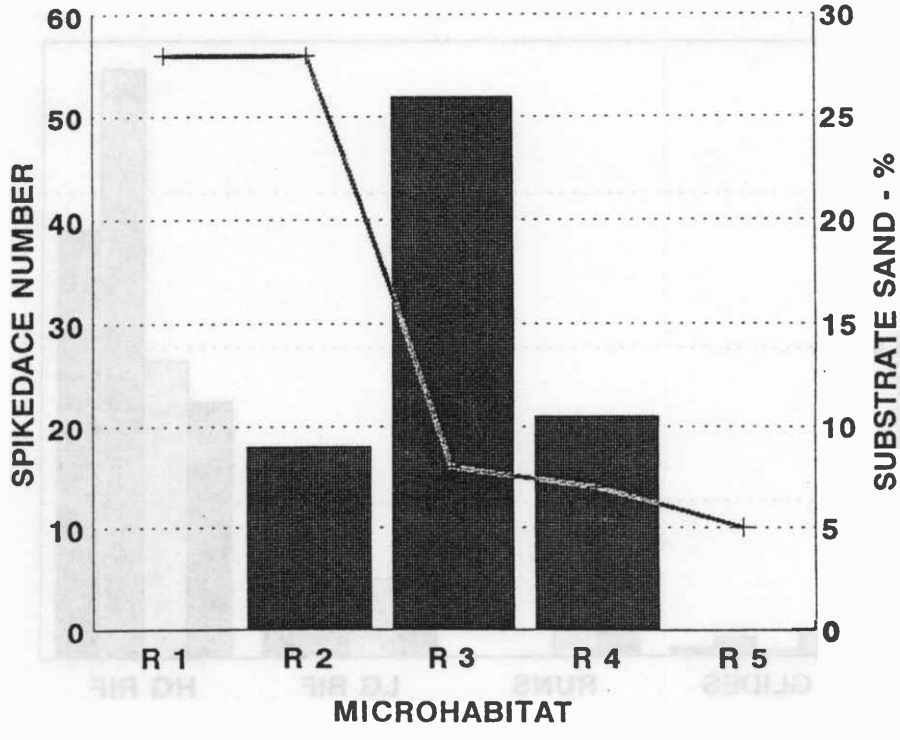

Figure 4. Spikedace numbers and percent sand in streambed substrate, Burnt Ranch study section, upper Verde River, March 1996.

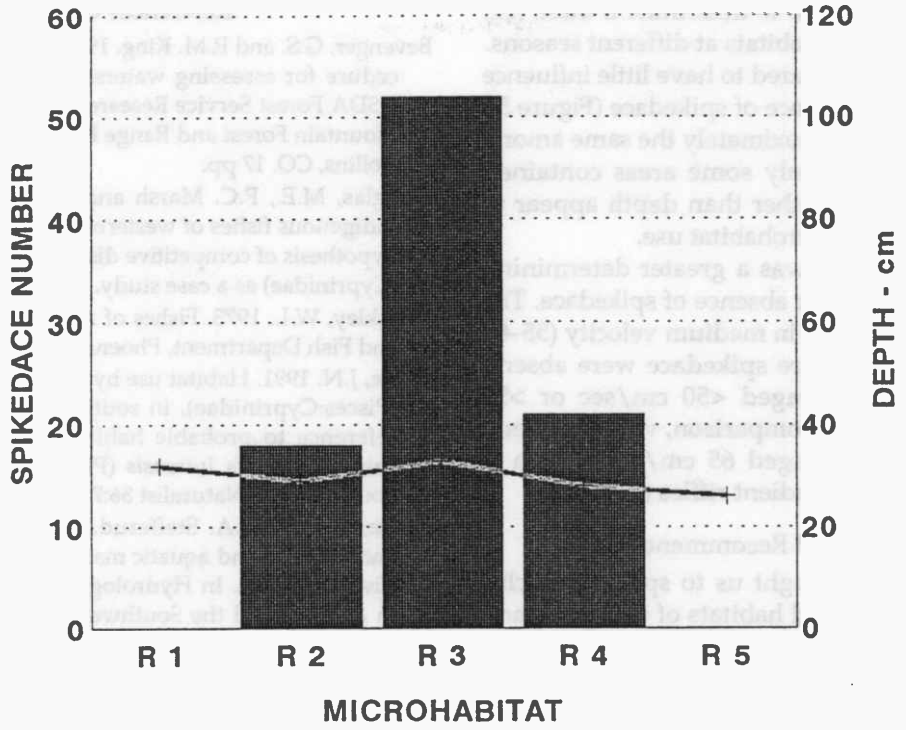

Figure 5. Spikedace numbers and stream depth, Burnt Ranch section, upper Verde River, March 1996.

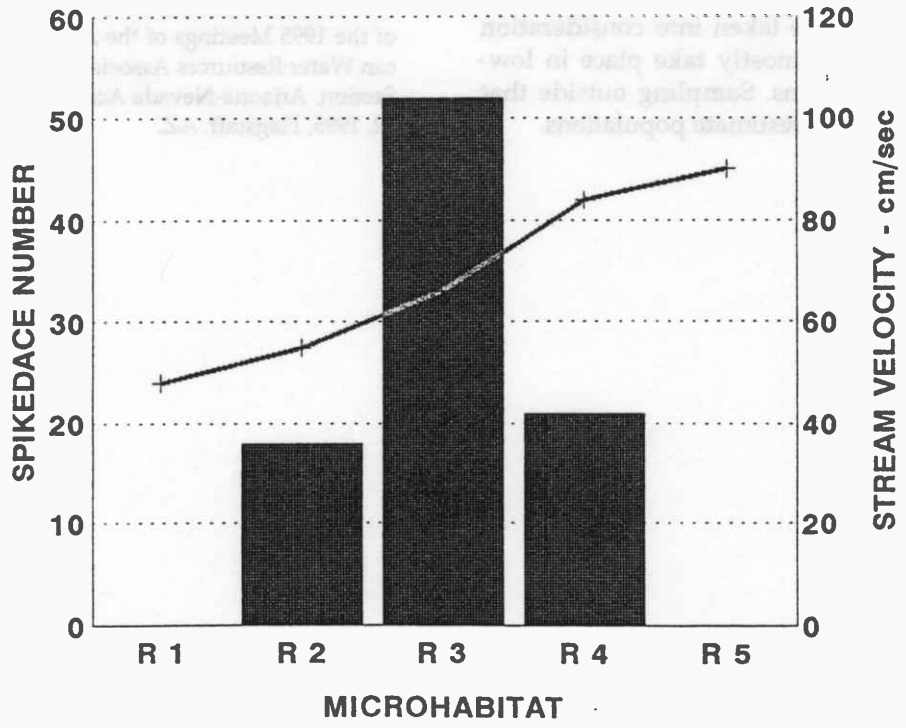

Figure 6. Spikedace numbers and stream average velocity, Burnt Ranch study section, upper Verde River, March 1996.

field sampling is needed to determine if these fish move into other microhabitats at different seasons.

Depth of water appeared to have little influence on the presence or absence of spikedace (Figure 5). That is, depth was approximately the same among all sample sites, yet only some areas contained spikedace. So factors other than depth appear to influence spikedace microhabitat use.

Velocity, however, was a greater determining factor in the presence or absence of spikedace. The species was found only in medium velocity (55–85 cm/sec) habitats. Where spikedace were absent, stream velocities averaged <50 cm/sec or >90 cm/sec (Figure 6). By comparison, where present, stream velocities averaged 65 cm/sec, which is characteristic of low-gradient riffles (Figure 2).

Conclusions and Recommendations

Briefly, our study brought us to specific conclusions as to the preferred habitats of the spikedace. Spikedace were found only in habitats with a specific range of velocity, substrate, and gradient characteristics. They were found in areas with velocities ranging from 55 to 85 cm/sec (mean 65), gradients of 0.4 to 0.6 percent (mean 0.5%) substrates of less than 10 percent sand, and in narrower channel sections.

For help in future sampling of the spikedace, all these factors should be taken into consideration and sampling should mostly take place in low-gradient riffles and runs. Sampling outside that range will greatly underestimate populations.

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SEDIMENT RATING CURVES FOR PINYON-JUNIPER WATERSHEDS IN NORTHERN ARIZONA

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Pinyon-juniper woodlands occupy large areas of the southwestern United States. Management of these ecosystems has been controversial, however, and their hydrology has been of particular concern because of past accelerated erosion (Gottfried et al. 1995). Approximately 1.4 million hectares (representing 35% of the pinyon-juniper woodlands on national forests in the southwestern region of the USDA Forest Service) have unsatisfactory soil and watershed conditions (USDA Forest Service 1993). This assessment has been based on evidence of surface erosion, gully erosion, and soil compaction, and on vegetative indices.

The decline in watershed condition has been attributed to increases in tree densities and corresponding decreases in understory plants (grasses, forbs, and half-shrubs), which provide protective cover on soil surfaces. Overstory trees were removed from many areas in the hope of remedying the situation by encouraging the production of understory plants. Another school of thought was that the observed erosion is a residual of past land-use practices, and causes of erosion are more complex (Gottfried et al. 1995).

The controversy over the hydrology and erosional dynamics in pinyon-juniper ecosystems has generated a number of research studies (Baker 1982; Gifford 1975; Wilcox 1994; Wright et al. 1976). As part of a general research effort to study the hydrologic and sediment-transport regimes of watersheds, the relative magnitudes of sediment exports are being analyzed in relation to watershed condition, streamflow-generation mechanisms, and management activities. Sediment rating curves, developed to describe relationships between the amount of sediment in suspension and streamflow discharge, can be used to estimate the effects of management activities on suspended sediment (Brooks et al. 1991).

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This paper reports upon the development of sediment rating curves for watersheds in pinyon-juniper woodlands of northern Arizona. It presents an analysis of suspended sediment concentration–streamflow discharge relationships for two pinyon-juniper watersheds subjected to vegetative conversion treatments, and a control watershed that represents untreated conditions. These comparisons provide a basis for determining the effects of the conversion treatments on sedimentation processes. Differences between snowmelt-runoff and convectional rainfall streamflow-generation events were also considered.

Study Area

The watersheds studied are located about 80 km south of Flagstaff on the Beaver Creek watershed, a tributary of the Verde River, in the Colorado Plateau physiographic province. These watersheds, established by the USDA Forest Service to evaluate effects of vegetative conversion treatments on multiple use values, supported stands of Utah juniper and pinyon. Descriptions of overstory species compositions and density conditions have been presented by Baker (1982), Clary et al. (1974), and Ffolliott and Thorud (1977) and will therefore not be detailed here.

The watersheds average about 1,600 m in elevation. Springerville very stoney clay soils derived from basalt and cinder parent materials predominate (Williams and Anderson 1967). The clays are primarily montmorillonite, which swell and shrink during each wet and dry cycle (Baker 1984). Infiltration rates for this soil series range between 2.0 and 6.4 cm/hr (Baker 1982). Stream channels on the watersheds have a southwesterly orientation. Annual precipitation averages 450 mm, occurring largely in two seasons. The most important precipitation from a streamflow-generating standpoint is that originating from the frontal storms of October through April, when about 60 percent of the annual precipitation falls. A second precipitation season is July through early September, when

high-intensity, short-duration, localized convectional storms are common. Average winter water yields (22.9 mm) account for 85 percent of the total annual water yield (Baker 1982). Suspended sediment accounts for approximately 75–80 percent of the total sediment discharge from the watersheds studied.

Vegetative Conversion Treatments

A cabling treatment was applied to a 131-ha watershed (WS1) in 1963. Larger trees were uprooted by a heavy cable pulled between two bulldozers. Smaller trees that were missed by cabling were chopped up by hand, slash was burned, and the watershed was seeded with a mixture of forage species. This treatment did not result in significant increases in annual water yields.

The second watershed was a 51-ha control (WS2) against which the cabling and herbicide treatments were evaluated. Conditions on this control watershed represented those obtained through custodial management; that is, through the use of minimal managerial inputs.

In a third treatment, a mixture of picloram (2.8 kg/ha) and 2,4-D (5.6 kg/ha) was applied by helicopter to 114 ha of a 147-ha watershed (WS3) in 1968; the remaining 33 ha were either not treated or the trees were sprayed with a backpack mist-blower. The intent of this treatment was to reduce transpiration losses by killing trees, reduce evaporation losses from the soil by leaving the dead trees standing, and reduce the amount of overland water flow trapped in the pits created when trees were uprooted by cabling (Baker 1984). The herbicide treatment resulted in an increase in annual water yields of about 160 percent (4.4 mm).

Analytical Procedures

A total of 191 paired suspended sediment concentration–streamflow discharge measurements (in excess of 0.05 m^3/s) obtained from 1975 through 1982 were used in deriving the sediment rating curves presented in this paper. Streamflow was measured in concrete trapezoidal flumes (Baker 1986). Either grab samples or integrated samples obtained with a DH-48 were analyzed by filtration to determine suspended sediment concentrations. When a sample of suspended sediment was collected, the time was indicated on a digital tape on continuous water-level recorders at the gauging sites. The sediment data used in this analysis were collected at time intervals greater than 1 hour to avoid the possibility of correlation among the paired data sets.

The following two types of events served as the basis for studying the effects of streamflow-generation mechanisms on suspended sediment concentrations:

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

Type 2. High-intensity, relatively short duration, mostly convectional rainfall events; rapid response time to peak streamflow discharge; streamflow duration of hours or a few days; occurs in late summer–early fall.

Rain-on-snow streamflow events, which represented less than 10 percent of the streamflow-generation mechanisms on the watersheds studied, were excluded from the analysis.

Sediment rating curves most frequently take a power function form, such as $C = aQ^b$, where C = suspended sediment concentration (mg/L), Q = streamflow discharge (m^3/s), and a, b = constants for a particular stream.

Sediment rating curves to be derived in a power function form are commonly approximated by least-squares linear regressions of logarithmic-transformed data (Walling 1977); therefore, this transformation was used to develop the sediment rating curves in this study.

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

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

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

Results and Discussion

A summary of statistics for streamflow discharge and suspended sediment concentration is shown in Table 1. Analysis of streamflow discharge–suspended sediment relationships began with the complete data sets from the watersheds studied. These data sets were subsequently partitioned into type of streamflow-generation mechanisms (snowmelt-runoff or convectional rainfall). This second step, the "Chow test" (see Kmenta 1986), was performed to test the null hypothesis that the parameters of the curves (a and b) had not changed significantly at the 5 percent level of significance.

Table 1. Summary of statistics for streamflow and suspended sediment concentration.

Event Type[1]	n[2]	Steamflow (m^3/s) Mean	StDv	Min	Max	Sediment Concentration (mg/L) Mean	StDv	Min	Max
Cabled Watershed									
All	117	0.102	0.056	0.058	0.261	8.88	13.33	0.39	73.07
1	104	0.101	0.056	0.058	0.261	8.97	13.64	0.38	73.07
2	12	0.104	0.030	0.058	0.127	5.50	5.38	1.17	15.99
Control Watershed									
All	32	0.093	0.035	0.058	0.216	7.87	6.94	2.10	40.17
1	21	0.096	0.030	0.060	0.216	7.74	8.21	2.10	40.17
2	10	0.073	0.011	0.058	0.084	7.25	2.59	4.29	12.01
Herbicide-Treated Watershed									
All[3]	42	0.088	0.028	0.059	0.151	10.55	13.69	0.74	80.17
1	41	0.087	0.027	0.058	0.151	10.18	13.64	0.74	80.17

[1]All = complete data set for each watershed; 1 = snowmelt–runoff events, 2 = convectional rainfall events.
[2]Samples (n) for the streamflow-generation mechanisms do not add up to the n for the complete data set for each watershed studied because of the inclusion of one frontal rainfall event occurring in late fall in the complete data sets.
[3]Event type 2 is not represented in the data set for herbicide-treated watershed.

Table 2. Sediment rating curve parameters with the 95 percent confidence limits, standard errors, coefficients of determination, and F statistic.

Event Type[1]	n[2]	a	95 Percent Confidence Limits[3]	b	95 Percent Confidence Limits	Fitted Error	R^2	F
Cabled Watershed (WS1)								
All	117	639.04	421.70–968.28	2.10	1.77–2.44	0.23	0.58	158.32
1	104	697.71	448.95–1084.17	2.13	1.77–2.48	0.23	0.57	140.17
2	12	356.71	55.41–2296.24	1.98	0.54–3.43	0.29	0.43	9.42
Control Watershed (WS2)								
All	32	107.26	53.63–214.53	1.17	0.58–1.76	0.18	0.33	16.36
1	21	196.01	69.34–554.09	1.48	0.63–2.34	0.22	0.38	13.15
2	10	616.84	220.64–1724.20	1.71	0.63–2.79	0.05	0.58	13.41
Herbicide-Treated Watershed (WS3)								
All[4]	42	435.75	135.94–1397.76	1.74	0.70–2.27	0.38	0.21	11.58
1	41	344.52	103.53–1146.69	1.65	0.57–2.72	0.39	0.18	9.59

[1]All = complete data set for each watershed; 1 = snowmelt-runoff events, 2 = convectional rainfall events.
[2]Samples (n) for the streamflow-generation mechanisms do not add up to the n for the complete data set for each watershed studied because of the inclusion of one frontal rainfall event occurring in late fall in the complete data sets.
[3]Confidence limits have been retransformed to original units for interpretation.
[4]Event type 2 is not represented in the data set for herbicide-treated watershed.

Measurements made during periods of high-streamflow discharges (Type 2 events) were assigned the same weight as measurements made during low-streamflow discharges (Type 1 events) in deriving sediment rating curves for each watershed. The parameters a and b of the relationships are presented in Table 2 with the 95 percent confidence limits, fitted standard errors, coefficients of determination, and F statistic.

There was a difference in the a and b values between the sediment rating curve for the cabled watershed (WS1) and that for the untreated watershed (WS2) at streamflow discharges greater than 0.15 m^3/s. This difference indicates that for streamflow discharges greater than 0.15 m^3/s, there are higher suspended sediment concentrations from the cabled watershed than from the control. These higher concentrations likely reflect the soil disturbances caused by uprooting trees in the cabling treatment (Clary et al. 1974). However, there was no statistical difference between the a and b values for the sediment rating curve derived for the watershed treated with herbicides (WS3), which experienced little soil disturbance as a result of treatment, and the untreated watershed.

Sediment rating curves were derived to represent suspended sediment responses to the type of streamflow-generation mechanisms. The parameters a and b of the resulting relationships are also shown in Table 2. There were no consistent differences in the a and b values of sediment rating curves when the data sets used to derive the equations were partitioned according to streamflow-generation mechanisms. This finding differs from that reported for a higher elevation ponderosa pine watershed, where a significant increase in the a and b values occurred when the data sets used to derive sediment rating curves were subdivided by streamflow-generation mechanisms (Lopes and Ffolliott 1993).

The highest suspended sediment concentrations were observed from the high-intensity, relatively short duration, convectional rainfall events on the cabled watershed. The lowest concentrations were associated with snowmelt-runoff events on all watersheds, probably because of a reduced flow energy and volume. However, over 50 percent of the annual sediment yields from the untreated watershed were generated during the winter period (Clary et al. 1974); this is likely because maximum streamflows in the period of record (Table 1) were greater in the winter than summer, and therefore would have more energy to transport sediments.

There was a significant difference in the suspended sediment concentration–streamflow relationship when the cabled and control watersheds are compared. Soil disturbance caused by the uprooting of trees on WS1 is the likely reason for this change. Sedimentation parameters can also change, in general, because of increased streamflows, but that did not occur after this treatment. The difference in results can be attributed to the fact that finer sediments (silt and clay) require less energy to remain in suspension and, as a consequence, can be transported by smaller flows than would be needed to move larger particles. This result, and those of Baker (1982) and Clary et al. (1974), are different than might be expected from the hypothesis that removal of trees to increase the production of herbaceous plants (forming a protective cover) will result in less erosion. No changes in sediment concentration–streamflow discharge relationships occurred on WS3 when herbicides were applied in a conversion treatment. The soil surface on this watershed was not disturbed by the treatment, since most of the herbicide was applied by helicopter.

Conclusions

Sediment rating curves have been developed for three pinyon-juniper watersheds in northern Arizona that had received different treatments from custodial management to vegetative conversion by cabling and application of herbicides. The cabling treatment resulted in increased suspended sediments at specified streamflow discharges because of the soil disturbances caused by the uprooting of trees, while the herbicide treatment did not cause a change. This finding could be helpful to managers in responding to questions about the hydrology and erosional and sedimentation dynamics on watersheds with similar soils, precipitation regimes, and vegetative conversion treatments.

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ROLE OF SNOW IN THE HYDROLOGY OF THE MADREAN PROVINCE IN SOUTHEASTERN ARIZONA

Peter F. Ffolliott,[1] *Gerald J. Gottfried,*[2] *and Malchus B. Baker, Jr.*[2]

In characterizing the hydrology of the Madrean Province of the southwestern United States and northern Mexico, Baker et al. (1995) stated that winter precipitation in the region is generally rain, although snow occasionally falls and accumulates in the higher elevations. In theory, snowmelt from these intermittent accumulations could contribute to the surface runoff originating on these mountain tops. Therefore, snowpack measurements taken on two snow courses in the Santa Catalina Mountains, near Tucson, have been analyzed in terms of water equivalents at peak seasonal accumulations. Estimates of snowpack densities calculated from measurements of snow depth and water equivalents are also presented. Variations from long-term average conditions are described. Inferences on the role of snow in the hydrology of the Madrean Province are made in relation to forest management activities to increase snowmelt water yields.

Snow Courses

The two snow courses, Bear Wallow and Rose Canyon, were established in the winter of 1947–48 as part of the Arizona Cooperative Snow Survey, a joint effort of the USDA Soil Conservation Service (currently the Natural Resources Conservation Service), the USDA Forest Service, the Salt River Valley Users Association, the Arizona Agricultural Experiment Station, and others. The purpose of the cooperative snow survey, which is conducted throughout the western United States, is to provide an index of snowpack water equivalents to forecast future snowmelt-runoff volumes, not necessarily to report the average snowpack conditions of the watershed in question. Therefore, the survey snow courses are established on sites that have a deep snowpack even in the drier years. With the exception of two aerial markers in the Pinaleno Mountains, the two snow courses on the Santa Catalinas were the only representatives of the survey in southeastern Arizona.

The snow courses were located in old-growth ponderosa pine forest stands composed mainly of trees 20 inches in diameter and larger. The overstory on the Bear Wallow course averaged 100 square feet of basal area per acre, while that on the Rose Canyon course averaged about 50 square feet of basal area per acre. There were insignificant numbers of understory trees on the two sites. The Bear Wallow course was 8,100 feet in elevation, on a generally northeastwardly aspect of 30–40 percent slopes; the Rose Canyon course was 7,300 feet in elevation, on a northwestwardly aspect of 30–35 percent slopes.

Transects of 10-foot pipes located at 50-foot intervals marked the sampling points on the snow courses on both sites. Eight sampling points comprised the Bear Wallow course and there were seven sampling points on the Rose Canyon course.

Collection of Data

Measurements taken in the survey have been used to estimate water equivalents and the condition of snow throughout the winter period of snowpack accumulation and melt; therefore, measurements made in these periods normally included peak seasonal snowpack accumulations. Snow depth and water equivalents were measured at the sampling points on the Bear Wallow and Rose Canyon courses with a Federal snow sampler. Water equivalents were determined by weighing the tube and snow with a spring balance.

Measurements were generally taken at about two-week intervals throughout the period of late January to the first of April from 1947–48 to 1982–83, when the measurements were discontinued. Reported snow course data represented the averages for all sampling points on the course (USDA Soil Conservation Service 1981).

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Results and Discussion

Snow was measured on the two courses at least once in each of the 36 years of record with the exception of the Rose Canyon course in 1972. The two courses were also snow-free during parts of most winter sampling periods. Snowpack accumulations on the higher Bear Wallow course tended to melt more slowly than those at Rose Canyon; and, the accumulations at Bear Wallow were augmented more frequently by storms throughout the winter period. The snowpacks were more intermittent and discontinuous on the Rose Canyon course, where the entire accumulation often melted between measurement dates.

The time of peak seasonal snowpack accumulations on the courses was variable, although peak water equivalents were most frequently measured on the February 15 sampling date. The greatest water equivalents were measured on February 15, 1966, with 19.6 inches at Bear Wallow and 13.6 inches at Rose Canyon. The average water equivalent (with standard error) at peak seasonal accumulations for the years of record was 6.1 ± 0.8 inches at Bear Wallow and 3.2 ± 0.5 inches at Rose Canyon. However, there was considerable variation in the annual values of peak seasonal water equivalents on the two courses (Figure 1). Typical of many meteorological and hydrologic data in the southwestern United States, these values rarely approached the long-term average; a few values were indicative of extreme highs, and a relatively large number of values were below the long-term average.

Estimates of snowpack densities were obtained from the measurements of total snow depth and water equivalents at the sampling points. No attempt was made to differentiate newly fallen from the residual snowpack in these samples; therefore,

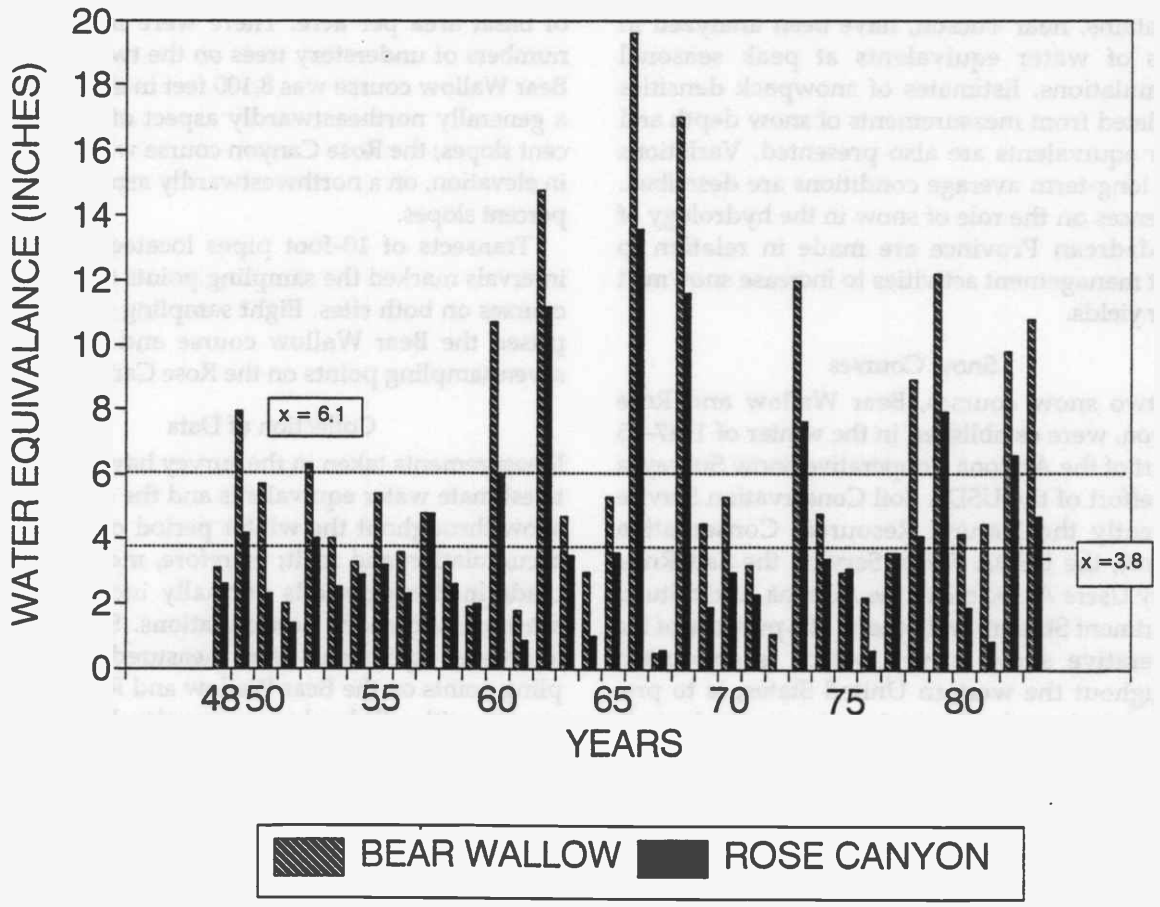

Figure 1. Annual values of peak seasonal water equivalents on the Bear Wallow and Rose Canyon snow courses. The respective average water equivalents at peak seasonal accumulations for the 36 years of record are indicated by the broken line.

the estimates of snow densities reflect values for the integrated snowpack at the time of measurement. Snowpack densities ranged from 0.15 to 0.47 gm cm^{-3} at Bear Wallow, and from 0.10 to 0.49 gm cm^{-3} at Rose Canyon for the period of record. The lower values generally represented newly fallen snow with little residual snowpack, and the higher values indicated a snowpack that probably was melting. The snowpack densities observed on the two courses and the temporal variations in the snowpack density values were similar to those reported for snow conditions of northern Arizona (Ffolliott 1985).

Snowpack densities at the time of peak seasonal accumulation were also variable, ranging from lows of about 0.15 to highs of almost 0.60 gm cm^{-3} throughout the 36 years of record. It was concluded, therefore, that the condition (ripeness) of the snowpacks at peak seasonal accumulation was variable in terms of ripeness or readiness to melt.

Management Inferences

Snow falling on high-elevation forests is a source of water throughout much of the southwestern United States. It might follow, therefore, that snowmelt water yields could be increased by modifying the structure of the forests through thinning, clearing, or combinations thereof. While these possibilities do occur elsewhere in the mountains of the southwestern United States (Ffolliott et al. 1989), they are essentially nonexistent in southeastern Arizona.

Snowpacks in the Madrean Province are largely intermittent, relatively shallow, and limited in their aerial extent; as a consequence, they do not represent a significant water supply potential in the region. Furthermore, high transmission losses in the stream channels suggest that only a fraction of any increases in water flows occurring as a result of modifying the structure of the forests would reach downstream places of use.

Discontinuities in forest cover on the mountain tops in the Madrean Province preclude implementation of management activities on a scale large enough to have a significant influence on the hydrologic regimes. It must be inferred, therefore, that the importance of snow as a source of water, and its role in the hydrology of the Madrean Province, is largely insignificant.

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FIRE SEVERITY AND WATERSHED RESOURCE RESPONSES IN THE SOUTHWEST

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A conceptual model outlining the effect of different fire severities on postfire hydrologic responses has been presented earlier (DeBano et al 1995). This conceptual model portrays fire severity as a continuum, ranging from minor resource responses under a cool-burning prescribed fire to the major responses that could be expected to occur during stand-replacing wildfires in forests. The fire response continuum in the southwestern United States is large (Baker 1990). In this model, prescribed fire conditions represent lower temperature–higher humidity burning conditions where fuel loading is minimal and fuel moisture is high. These conditions produce lower fire intensities and thus expose the soil and water resource to lower fire severities. Prescribed fire usually has minor hydrologic impacts on watersheds because the surface vegetation, litter, and forest floor are only partially burned (Baker 1990). Other resources (soils, wildlife, etc.) are also changed little by a prescribed fire. On the other end of the fire response continuum, fire behavior more nearly represents that present during wildfires, where the temperatures, wind speeds, and fuel loadings are high and the humidities and fuel moistures are low. In contrast to prescribed burning, wildfires can have a major effect on basic hydrologic processes, leading to increased sensitivity of the site to erosion and to reduced land stability (Baker 1990). Large changes also occur in other resources, including denuded landscapes, large losses of plant nutrients, and so forth.

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

Before the conceptual model described above can be developed, response functions for the different resources to a range of fire severities need to be better quantified. Other important factors affecting the postfire hydrologic scenario are postfire precipitation patterns and slope. The hydrologic response model is generally visualized as a three-dimensional surface with a particular hydrologic response (peak flow, sedimentation rate, etc.) being a function of both fire severity and a time variable reflecting climatic events following a fire. Individual differences in site parameters unique to a given area (e.g. slope, aspect, soils, etc.) would also have to be considered. The immediate soil and watershed response (how much litter and plant cover had been destroyed by the fire, amount of nitrogen volatilized, etc.) would be most closely related to fire severity, and would probably be a non-linear function such as the one that describes infiltration into a wettable dry soil. An additional time function, reflecting precipitation events, would be necessary to define the longer term hydrologic responses to a particular fire severity. The dimension of time is essential for the model because of the possibility of variable precipitation events that could follow a fire. For example, a low-intensity prescribed fire can produce substantial runoff and soil loss as sediment if

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the fire is immediately followed by high-intensity rainstorm events. Conversely, severely burned watersheds can produce little runoff and erosion if a fire is followed by a relatively mild year with gentle rains and warm temperatures that allow a protective vegetation cover to develop. The pattern of precipitation events over time is stochastic in nature, and would have to be viewed in a probability framework so that relevant outcomes could be explored. The effects of "big" storms in the Southwest are of particular importance. It may be that low-probability, high-impact events are more important in prescribed burning decision criteria than long-term average outcomes, reflected in the expected values of probability distributions. Information for the two-dimensional part of this model has been developed by Ffolliott et al. (1988), and with some modification could be used as a starting point for developing the time dimension following fire.

The first iteration of the above model is being developed to describe select watershed responses. All available information on hydrologic responses for the southwestern United States is currently being consolidated to define and quantify hydrologic responses to wildfires and, more importantly, to cooler burning prescribed fires. Unfortunately, only a meager data base is available on the hydrologic responses of watersheds burned at lower severities. The best data set available was collected during a case study of the Rattle Burn in north-central Arizona (Campbell et al. 1977), and it was used to quantify the conceptual model described above for ponderosa pine forests. Additional information is also presented on other vegetation types, but it only reflects differences between wildfires and prescribed burns and not intermediate fire severities.

Ponderosa Pine Example

A wildfire designated as the Rattle Burn (May 7–9, 1972) swept through 290 ha of even-aged stands of ponderosa pine growing on sedimentary-derived soils on the West Fork of Oak Creek of the Coconino National Forest, at an elevation of 2,040 m about 29 km southwest of Flagstaff. Three small watersheds, representing severe and moderate burns and an unburned control, were established to assess the effects of wildfire on hydrology, soils, timber and forage production, and wildlife populations. On the moderately burned watershed, fire was generally confined to the forest floor. The intensity on the moderately burned site was calculated to be 9,000 kjoules/sec/m. Most of the trees were killed on the severely burned watershed. The fire intensity on the severely burned watershed was calculated to be 35,000 kjoules/sec/m. A nearby unburned watershed served as a control.

Streamflow was measured with automatic stream stage recorders at the outflow points of each watershed between 1973 and 1975. The mean annual water yields were 0.5 cm from the unburned watershed, and 2.0 and 2.8 cm from the moderately and severely burned watersheds, respectively. Although the differences in annual water yield were small between the watersheds, the differences in peak flows were much larger. The highest annual peak discharge on the control watershed was only 0.01 $m^3/s/km^2$. On the moderately burned watershed, peak discharge reached 0.24 $m^3/s/km^2$, while on the severely burned watershed the peak discharge exceeded 4.067 $m^3/s/km^2$. The number of runoff events during the period 1972–75 also increased with severity of burning, ranging from 6, 15, and 25 for the unburned, moderate, and severe burns, respectively. The increased peak runoff was also reflected in suspended sediment yields. The total suspended sediment yield for 1972–75 was 3, 20, and 1559 kg/ha for the unburned, moderate, and severe burns, respectively. The suspended sediment losses occurred during the first two years, and particularly during the second year when all-time precipitation records were set.

The suspended sediment data from the Rattle Burn were used to establish maximum responses that might be expected over time from a ponderosa pine forest that had been subjected to different burning severities in situations similar to those on the Rattle Burn. Best estimates of the duration of sediment production response were combined with the maximum numbers to generate a response relationship for a particular severity (Figure 1). It is assumed, in general, that annual sediment production increases on watersheds burned at moderate severity will be less in magnitude and much shorter in duration than those that were severely burned. Probably after a couple of growing seasons, little response will be evident on the moderately burned watersheds. In contrast, the response curves for watersheds receiving severe burns will be greater in magnitude and longer in duration than for the moderately burned watersheds, perhaps lasting up to 15 years. These time periods are consistent with other resource responses (Ffolliott et al. 1988). Although no information was available on the low-severity fire, it was assumed that there would be no substantial

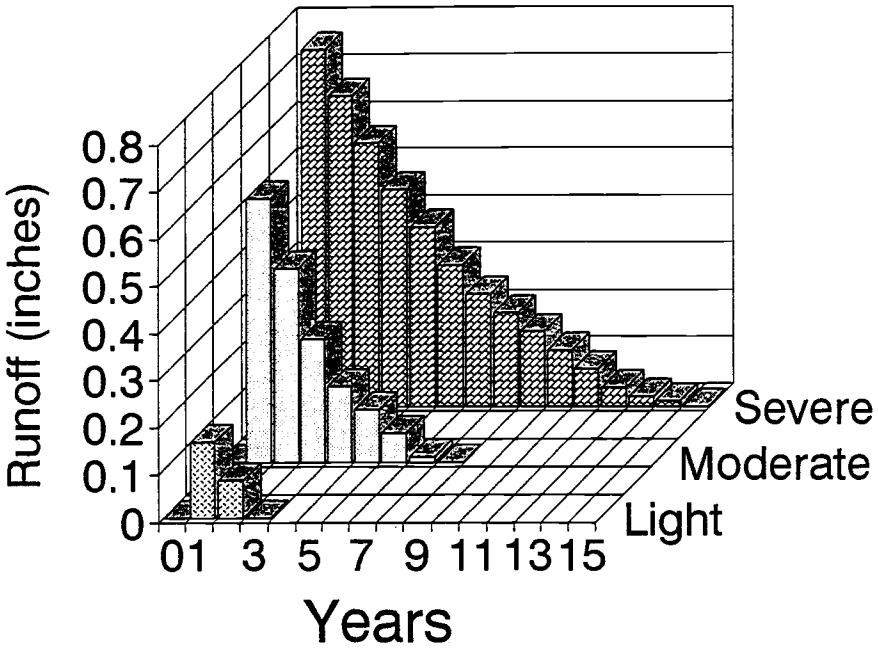

Figure 1. Average increases in annual sediment yield over time from ponderosa pine watersheds that were severely, moderately, and lightly burned.

increases in streamflow or sediment production. Data from ponderosa pine stands in California confirm that no significant increases in sediment losses can be expected following a low-intensity prescribed burn (Biswell and Schultz 1965).

Chaparral and Other Southwestern Vegetation Types

Data showing the hydrologic response to fire for chaparral and other vegetation types in the Southwest are limited to several case studies reporting on wildfires and some prescribed burns. There are only a few published reports on watershed responses to prescribed burning. Unfortunately, data do not exist for many vegetative types on intermediate fire severities, nor are there studies on paired watersheds such as were available for the ponderosa pine example discussed above. Also, the differences in prescribed burn severities between wildfires and prescribed burns in chaparral are usually not as great as for ponderosa pine forests, because chaparral fires are carried mainly by the brush canopies and consequently the severities are high in those areas where the canopy is consumed. As indicated above, the overall severity for a watershed may be mitigated by burning in a mosaic pattern where the entire area is not burned.

Fire increases sediment loss in all vegetation types in the Southwest, as well as in the ponderosa pine forests discussed earlier. Table 1 shows substantial differences in sediment losses from wildfires compared to prescribed burning in both Arizona and California chaparral. In California chaparral, for example, sediment losses after wildfires amounted to 55.3 Mg/ha following wildfires, compared to losses of 5.5 Mg/ha from similar unburned sites (Krammes 1960). After wildfires in Arizona chaparral, sediment losses up to 204 Mg/ha have been reported (Glendening et al. 1961). The available data indicate that sediment losses following prescribed burns in chaparral are much lower than for wildfires. For example, a study by Pase and Lindenmuth (1971) in central Arizona showed sediment losses of only 3.8 Mg/ha after a prescribed burn. Likewise, sediment yields from the Battle Flat watersheds following a mosaic burn amounted to about 0.03 Mg/ha compared to only a trace of erosion (less than .001 Mg/ha) from a nearby watershed (Overby and Baker 1995). In general, the sediment losses following chaparral fires (both wild and prescribed) usually exceeded those reported for ponderosa pine forests (Table 1).

Slope also has an important effect on the sediment losses following both prescribed burns and wildfires, and needs to be considered when using the conceptual model described above. Some information is available on the effect of fire on

Table 1. Sediment losses the first year after prescribed burns and wildfires in the Southwest.

Location	Watershed Condition		Sediment Loss First Year (Mg/ha)	References
California	Ponderosa pine	Control	<0.001	Biswell and Schultz 1965
		Prescribed Fire	<0.001	
California	Chaparral	Control	0.043	Wright et al. 1976
		Wildfire	28.605	
California	Chaparral	Control	5.530	Krammes 1960
		Wildfire	55.300	
Arizona	Chaparral	Control	0.000	Pase and Lindenmuth 1971
		Prescribed Fire	3.778	
Arizona	Chaparral	Control	0.175	Glendening et al. 1961
		Wildfire	204.000	
Arizona	Ponderosa pine	Control	0.003	Campbell et al. 1977
		Wildfire	1.254	
Arizona	Mixed conifer	Control	<0.001	Hendricks and Johnson 1944
		Wildfire, 43% slope	71.680	
		Wildfire, 66% slope	201.600	
		Wildfire, 78% slope	369.600	
Texas	Juniper\grass	Control, level	0.025	Wright et al. 1982
		Burned, level	0.029	
		Control, mod. slope	0.076	
		Burned, mod. slope (15–20%)	1.874	
		Control, steep slope	0.013	
		Burned, steep slope (43–54%)	8.443	

different slopes in pinyon-juniper woodlands and mixed conifer forests. In mixed conifer forests in central Arizona, sediment losses the first year following a wildfire were 0.001 Mg/ha on level topography compared to up to 370 Mg/ha on a 78 percent slope (Hendricks and Johnson 1944, Table 1). Intermediate amounts of 72 and 202 Mg/ha occurred on 43 and 66 percent slopes, respectively. Similar effects of slope on sediment losses have been reported for juniper/grass-covered watersheds in Texas (Wright et al. 1982). Sediment losses increased nearly 650 fold on slopes of 43–54 percent compared to trace amounts on level ground. Increases of about 25 fold were reported on slopes of 15–20 percent.

Conclusions

Fire occurs frequently in southwestern ecosystems in the form of either wildfires or managed prescribed burns. Managers are often asked to assess the effects of fire on the soil and water resources. It is important during these evaluations to clearly distinguish between wildfires and prescribed fires, and the severities associated with each. A conceptual model outlining the different responses is presented in this paper. The model portrays fire severity as a continuum, with responses ranging from minor resource responses under a cool-burning prescribed fire to major responses which could be expected to occur during stand-replacing wildfires in forests; the fire response continuum in the southwestern United States is large (Baker 1990). In this model, prescribed fire conditions represent lower temperature–higher humidity burning conditions where fuel loading is minimal and fuel moisture is high. These cooler burning conditions produce lower fire intensities, and thus expose the soil and water resource to lower fire severities.

Postfire precipitation regimes and slope are also important factors when evaluating hydrologic responses to fire. The effects of "big" storms in the Southwest are of particular importance. Big storms on gentle slopes that have burned less severely could yield large amounts of sediment. Conversely, even severely burned steep slopes may not

contribute substantial amounts of sediment if only low-intensity storms occur after the fire until vegetation becomes established. It may be that low-probability, high-impact events are more important in prescribed burning decision criteria than long-term average outcomes, reflected in the expected values of probability distributions. These postfire responses to a given precipitation regime will also depend on the slope of the burned area. Sediment production generally increases exponentially with slope and this increase can be magnified by fire.

Although the conceptual model of a fire continuum is useful for describing general soil and watershed responses, more quantitative data are needed before it can be used to assess the magnitude of response. Some information is available in the literature for most vegetation types in the Southwest. However, these data are generally limited to the effect of wildfire on hydrologic responses. Only a few studies have reported hydrologic responses as they are related to fire severity. The best data available are for ponderosa pine forests in central Arizona, and this data set is limited to one case study.

Although the conceptual model cannot be fully implemented with existing data sets, an evaluation of the existing information clearly shows that low-severity prescribed fires in all vegetation types produce much smaller hydrologic responses (e.g. runoff, sediment losses) than are experienced during wildfires. In many cases, more than a hundred-fold increase in response has been reported for wildfires compared to prescribed burns. Because there are large differences in watershed responses between wildfires and prescribed burns, it is necessary to clearly differentiate between the two when assessing fire effects. For example, it is not of much value to use published information on wildfire effects when trying to predict the hydrologic responses of a planned prescribed burn due to the differences in fire severities and associated hydrologic responses. Generally, fire increases sediment losses, but this impact may be tempered by postfire precipitation regimes and slope.

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RECONSTRUCTION OF PRECIPITATION AND PDSI FROM TREE-RING CHRONOLOGIES DEVELOPED IN MOUNTAINS OF NEW MEXICO, USA AND SONORA, MEXICO

Jose Villanueva-Diaz and Guy R. McPherson[1]

Long quantitative records of year-by-year meteorological observations are limited or nonexistent in the southwestern United States and northern Mexico. Such records are invaluable to researchers studying paleoclimate dynamics and forcing functions that influence the climate system.

Trees store information on their annual rings about climates and their effects on the structure and composition of local forest ecosystems. Therefore, tree rings have been used as sources of information to investigate the role of natural and anthropogenic disturbances as well as environmental history as recorded by trees.

The ability to assign actual calendar dates to each ring of certain tree species allows for studies of the yearly climatic conditions necessary for growth. By comparing patterns of growth of climatically sensitive trees with patterns of environmental conditions for the same years, a reconstruction may be derived and applied to past years for which written records of environmental conditions do not exist (Fritts 1976; Conkey 1979).

Given the importance of understanding the influence of climate on forest dynamics, the objectives of this study were to use tree-ring series as proxy data for precipitation and PDSI reconstruction, and to assess the role of climate and land-use practices on forest dynamics.

Background of the Animas Mountains and the Sierra los Ajos

The Animas Mountains are located in southwestern New Mexico (31°35'N latitude, 108°47'W longitude). The highest peak rises to 2600 m, and the mountains cover an area of 100 sq m on the Gray Ranch in southern Hidalgo County (Wagner 1977). The Animas Mountains were formed by regional uplift and widespread volcanic activity during the Tertiary. During the Cretaceous, basalt rock was erupted, followed by granodiorite rock. In the late Tertiary volcanic activity was renewed with eruptions of rhyolite, tuffs, and basalt (Arras 1979; Stone and O'Brian 1990; Wagner 1977).

The Sierra los Ajos are located in Sonora, Mexico (30°55'N latitude, 109°55'W longitude), about 100 km southwest of the Animas Mountains. The main peak rises above 2600 m (COTECOCA 1973) (Figure 1). The Sierra los Ajos comprise a complex geologic formation characterized by a heterogeneous lithic composition. Outcrops from Precambrian and Holocene age are found in these mountains (Aponte 1974).

Both areas are characterized by dissected topography. Rocky to shallow soils are typical (Soil Conservation Service 1973; Garza-Salazar 1993). The Animas Mountains and Sierra los Ajos both have a bimodal precipitation pattern with about 60 percent of the average annual precipitation (450–750 mm depending on elevation) falling in July–September and 40 percent occurring in the winter months. Temperatures above 32°C are common during the summer months and usually range between 12°C and -5°C during the winter months. An average evaporation of 2340 mm can be expected for both mountain ranges (Tonne et al. 1992; Solis-Garza et al. 1993).

Few major drainages in the Animas Mountains extend beyond the edges of the mountains and they rarely carry water. Little permanent water is available with the exception of springs located in canyon bottoms and the lower parts of the mountains (Hubbard 1977; Wagner 1977; Fish and Wildlife Service 1989).

The Sierra los Ajos is part of three important watersheds: the Rio Sonora, the San Pedro, and the Bavispe. Streams rising in the Sierra los Ajos contribute to three major systems: the northward-flowing Rio San Pedro, the southward-flowing Rio Sonora, and the Rio Yaqui (Fishbein et al. 1995). Some of the intermittent or perennial flows rarely reach the major rivers of the valleys. The volume of water produced is important for some rivers of

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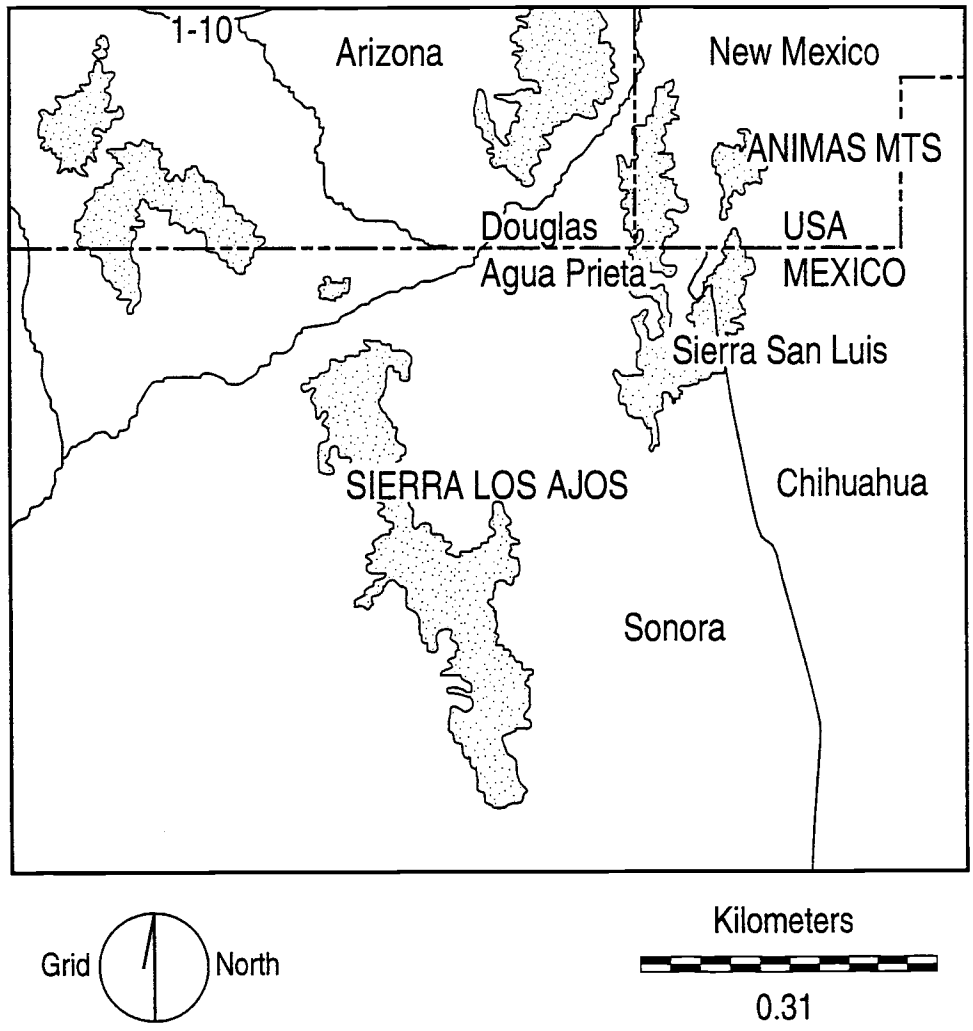

Figure 1. Geographical location of the Animas Mountains, New Mexico and the Sierra los Ajos, Sonora.

the region, such as the Rio Sonora, whose water is used for irrigation purposes in adjacent areas along the river. Some water of this river is confined in larger dams and later released for human use in cities, such as Hermosillo. Important riparian communities have been established along streams coming from the Sierra los Ajos.

The proximity of both the Animas Mountains and Sierra los Ajos to the Sierra Madre and the southern Rocky Mountains has resulted in a floristically diverse vegetation comprised of both northern and southern elements (Wagner 1977; Brown 1982; Fishbein et al. 1995).

Wagner (1977) categorized the vegetation of the Animas Mountains main massif into three basic types: lower encinal, upper encinal, and forests. The forest community is found between 1980 and 2600 m elevation (Hubbard 1977). Some of the major species present in this vegetation type are Douglas-fir (*Pseudotsuga menziesii*), found in canyons and on ridges of the northern portion of the mountain, and mixed with this species are southwestern white pine (*Pinus strobiformis*), ponderosa pine (*Pinus ponderosa* var. *Arizonica*), and Chihuahuan pine (*Pinus leiophylla* var. *chihuahuana*).

The mixed pine forest is found between 2300 and 2600 m elevation and is composed of ponderosa pine, southwestern white pine, Chihuahuan pine, and Apache pine (*Pinus engelmannii*) (Wagner 1977).

The pinyon pine-juniper woodland is found between 2300 and 2450 m and is composed of pinyon pine (*Pinus discolor*), alligator juniper (*Juniperus deppeana*), and oaks (*Quercus* spp.).

The Sierra los Ajos supports biotic communities classified as mixed conifer forest, montane mea-

dows, montane chaparral, oak woodland, and riparian forest (Garza Salazar 1993; Fishbein et al. 1995). The mixed conifer forest is restricted to north-facing slopes between 1900 and 2600 m in elevation. Northern aspects are dominated by Douglas-fir, associated with Gambel oak (*Quercus gambelii*) and madroño (*Arbutus arizonica*).

Methods

Permanent plots were established in each of four representative stands of forest communities in both mountain ranges, during 1992 and 1993. Increment cores were extracted from living trees. Cores were prepared and dated following standard dendrochronological techniques (Stokes and Smiley 1968; Swetnam et al. 1985).

To assign each tree to its exact year of formation, skeleton plots for each core were constructed and crossdated with a composite skeleton plot developed from Douglas-fir and ponderosa pine cores collected in each of the mountain ranges. Crossdating was verified and corrected with COFECHA (Holmes 1994).

The program ARSTAN (Cook 1985; Holmes 1994) was used to remove nonclimatic autocorrelation from each ring series. Standardization of tree-ring measurement series is necessary to remove growth trends produced by normal physiological tree aging, the effects of varying site productivity among different trees, and changes in the tree's environment that are unrelated to climate (Fritts 1976). Linear or negative-exponential single detrending was chosen to find the best fit for each series. Straight-line and negative-exponential methods are conservative in removing only monotonic trends. An index of tree growth for any year was obtained by dividing the actual ring width by that predicted from the curve. Relationships between tree-ring index chronologies and climate were investigated using response-function and correlation analysis (Fritts 1976; Guiot et al. 1982). To define those months or seasons in which precipitation and PDSI were significantly correlated with tree growth, regional climatic data developed for NOAA climatic divisions 09 and 07 for southwestern New Mexico and southeastern Arizona respectively over the period 1896–1993 were considered. Monthly total precipitation was seasonalized for the previous and current year seasons, since the tree integrates the effects of favorable or poor conditions over more than one growing season (Fritts 1976). Monthly PDSI, which integrates precipitation, temperature, soil conditions, sunlight duration, and geographic area into one drought index (Palmer 1965; Karl 1986), was also correlated with ring-width index.

To obtain transfer functions that predicted precipitation and PDSI as functions of tree growth, calibration equations were developed from the tree-ring index chronologies on subperiods (1896–1930 and 1931–1993) using least-square analysis. Calibrated equations were verified by testing performance on the subperiod withheld during calibration. The roles of the two subperiods were then reversed, and the model was calibrated with the subperiod. Predicted climate values were compared to actual climate values using well-established statistics (i.e., correlation analysis, reduction of error tests, nonparametric sign tests, and the product mean test) to determine how near the predicted values were to the actual climate values for the historical period (Fritts 1991).

Since the calibration-verification procedure for both subperiods proved to be a robust estimator of climate, the whole period (1896–1993) was considered to develop a new linear regression equation to reconstruct the selected climatic variables for the entire length of the tree-ring chronology. The use of the complete data period maximizes the number of observations, thus increasing the degrees of freedom for the final regression equation.

Short-term trends in climatic conditions between mountain ranges were analyzed. We fit 10-year smoothing splines through the reconstructed time series. Decadal-length climate episodes may indicate above and below normal climate periods (with respect to the reconstructed mean) that may then be related to stand dynamic processes (e.g., seedling recruitment, rate of tree growth, forest structure and composition, fire frequency).

Periods of above or below normal climate (precipitation and PDSI) were compared to other climate reconstructions generated for the southwestern United States in order to analyze the effect of a regional climate signal on tree growth.

The influences of climate and land-use practices (fire suppression, grazing, logging) were related to the establishment and growth of trees in both mountain ranges.

Results and Discussion

Correlation and response-function analysis for the Animas Mountains chronology indicated a significant response to winter precipitation extending from the previous October to the current January. The variance explained by seasonal precipitation was only 38.5 percent, which was greater than values obtained for other seasonalized periods.

The Sierra los Ajos seasonalized precipitation for previous July to current July explained 36 percent of the variance in cambial growth. A similar seasonal period for reconstruction of precipitation for northern New Mexico was used by Rose et al. (1981) and Grissino-Mayer (1995).

The amount of variance explained by the chronologies for these mountain ranges was relatively low compared to other chronologies developed in the southwestern region (e.g., El Malpais Long chronology in New Mexico, 57.4%; Jemez Mountains chronology in New Mexico, 63%) (Grissino-Mayer 1995; Touchan and Swetnam 1995). Differences in variance accounted for could be attributed to the lack of suitable meteorological data to relate tree growth and climatic variables, differences in land-use history for compared mountain ranges, or difference in length between chronologies.

Current July PDSI was significantly correlated with tree growth for both mountain ranges. In the Animas Mountains, cambial growth explained 41 percent of the variance in current July PDSI, whereas in the Sierra los Ajos, 45.2 percent of the variance was explained. Other studies in the southwestern United States have demonstrated that PDSI is usually associated with tree growth during the months of June, July, and August (Baisan and Swetnam 1990; Swetnam and Baisan, in press). As indicated by PDSI values, July apparently was the month during which availability of water was more limiting to cambial growth.

Calibration and verification models on subsets of years indicated that precipitation and PDSI reliably predicted tree growth in both mountain ranges. Therefore, new calibration models were developed for the entire period containing climatic data (1896–1993). We used these calibration models to estimate PDSI and precipitation in the Animas Mountains for the period 1760–1992, and PDSI and precipitation for the Sierra los Ajos for the period 1838–1992. Comparisons between actual and reconstructed data confirmed that the calibration models adequately simulated high frequency variability for both precipitation and PDSI in both mountain ranges (Figure 2).

Reconstructions revealed decadal trends in precipitation during the last 250 years at the Animas Mountains (Figure 3). Above-normal precipitation occurred during the periods 1783–1788, 1790–1800, 1849–1852, 1876–1880, 1905–1908, 1918–1933, and 1975–1992. Since 1900, above-normal precipitation has been related to El Niño episodes (i.e., 1915, 1919, 1926, 1958, 1983, and 1993), or decades of frequent El Niño events and expanded circumpolar vortex (1900–1930 and 1960–1993). The latter event is characterized by greater precipitation in fall, winter, and spring, with slightly reduced rainfall during the summer (Andrade and Sellers 1988; Betancourt et al. 1993). Below-normal precipitation occurs during La Niña events (i.e., 1904, 1917, 1925, 1943, 1950, 1955, 1974, and 1989) or decades when the circumpolar vortex contracts (Betancourt at al. 1993). Short-term periods of below-normal precipitation occurred during 1807–1813, 1817–1825, 1899–1904, 1934–1939, and 1947–1957. Some of these episodes have been detected in other southwestern precipitation reconstructions (D'Arrigo and Jacoby 1992; Fritts 1991; Grissino-Mayer 1995; Touchan and Swetnam 1995).

Reconstruction of annual precipitation in the Sierra los Ajos for the last 155 years indicated the presence of dry and wet episodes, coinciding with similar periods in the Animas Mountains and other southwestern United States precipitation reconstructions. Above-normal precipitation (425–457 mm) occurred during 1844–1849, 1856–1864, 1905–1924, and 1975–1991. Below normal precipitation was recorded for the periods 1839–1843, 1863–1864, 1871–1873, 1880–1890, 1899–1904, and 1946–1957.

Reconstructed July PDSI demonstrated a similar trend in both mountain ranges (Figure 4). In general, reconstructed PDSI revealed wet and dry spells that were matched with similar periods recorded by the reconstructed precipitation.

Before the 1900s seedling recruitment of Douglas-fir in the Animas Mountains apparently was enhanced by the presence of fires followed by above-normal precipitation. A similar pattern was observed for establishment of pines in the Sierra los Ajos, where the presence of fires followed by above-normal precipitation periods favored increased establishment of ponderosa pine seedlings. After the 1900s, increased establishment of Douglas-fir seedlings in the Animas Mountains may be attributed to a fire-suppression policy coupled with favorable climatic conditions.

The Sierra los Ajos has not experienced fire suppression, and the relationship between fire, precipitation, and seedling establishment has not changed, as evidenced by the age class distribution (Figure 5). The virtual absence of old ponderosa pine trees in the Sierra los Ajos could be attributed to logging activities but probably not to the effects of fires: ponderosa pine trees are very fire tolerant (Wright and Bailey 1982), and many individuals have survived several fires in this study.

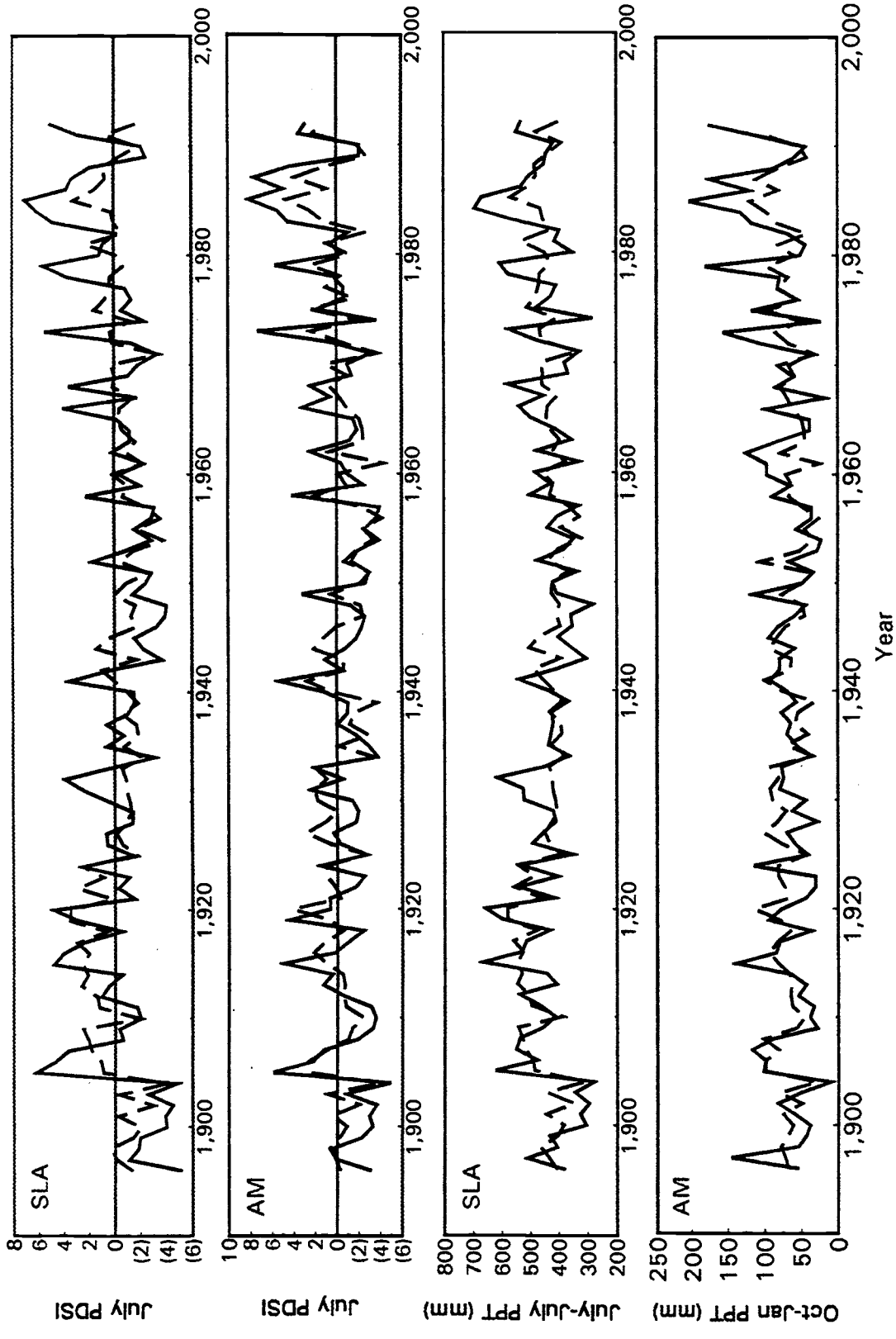

Figure 2. Comparisons between actual (solid line) and reconstructed (dashed line) precipitation and PDSI for the Animas Mountains, New Mexico and the Sierra los Ajos, Sonora, Mexico.

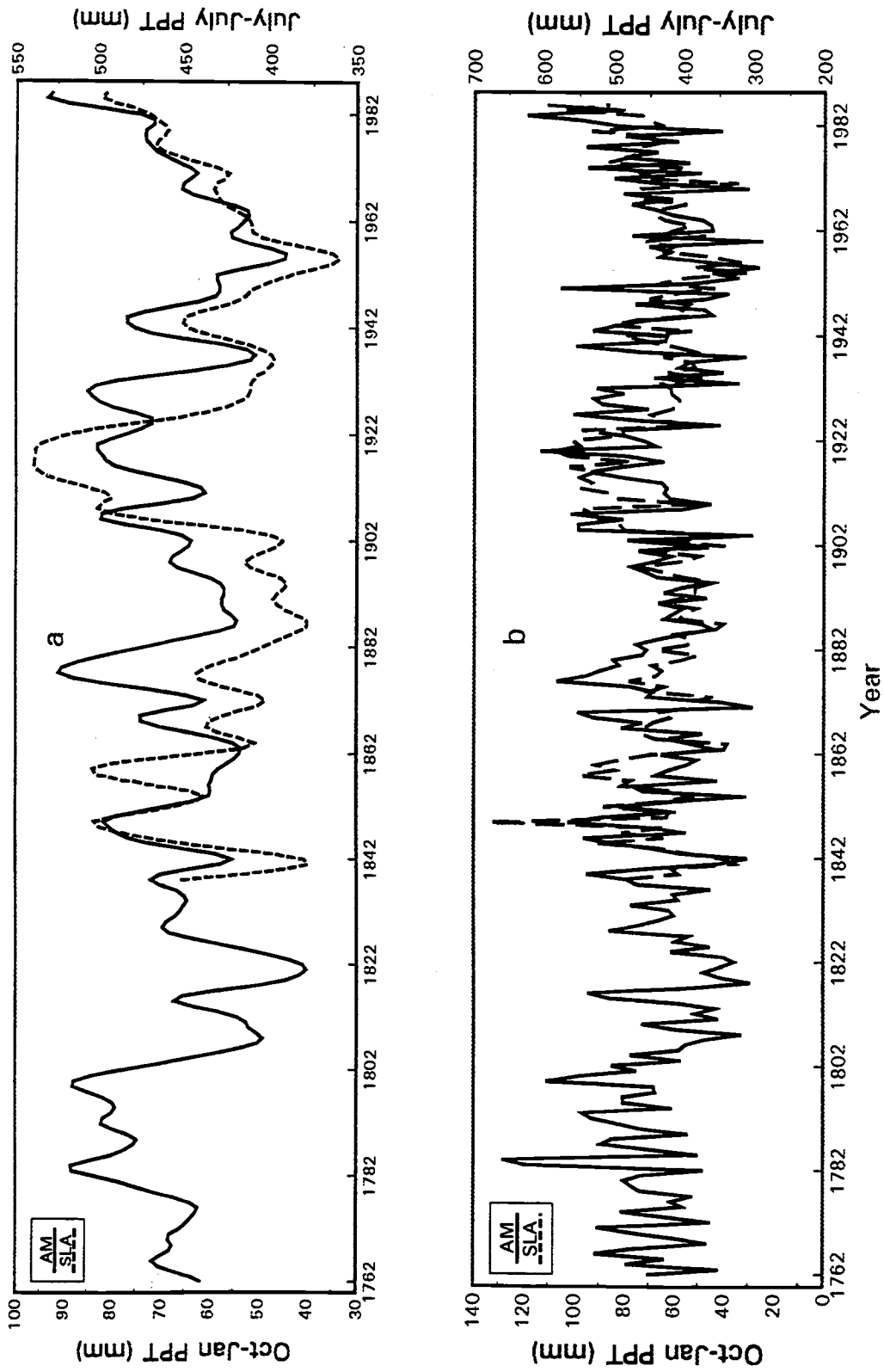

Figure 3. Reconstructed precipitation for mountain ranges using 10-year spline to emphasize decadal-scale precipitation differences (a) and annual precipitation fluctuations (b).

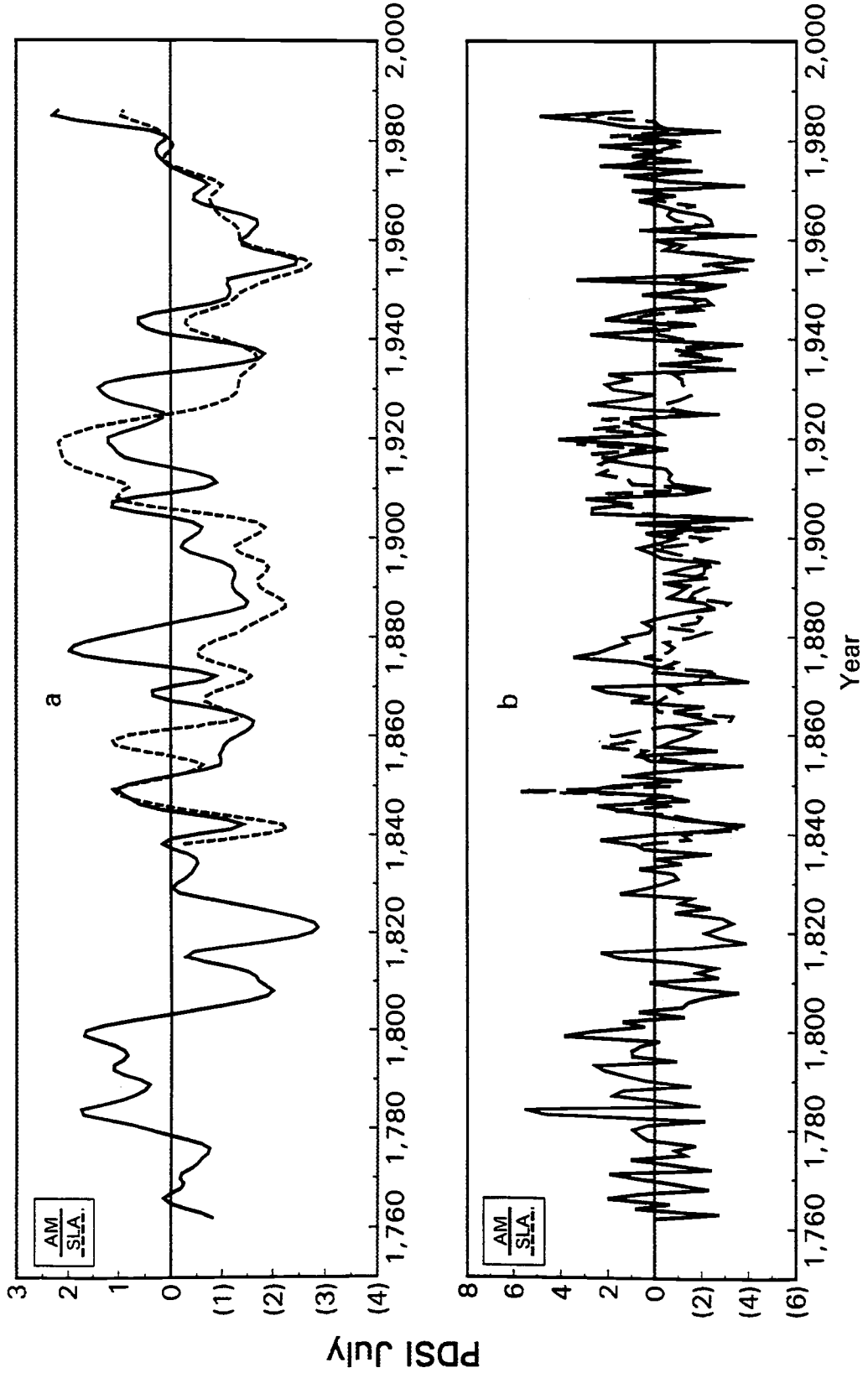

Figure 4. Reconstructed PDSI for mountain ranges using 10-year spline to emphasize decadal-scale PDSI differences (a) and annual PDSI fluctuations (b).

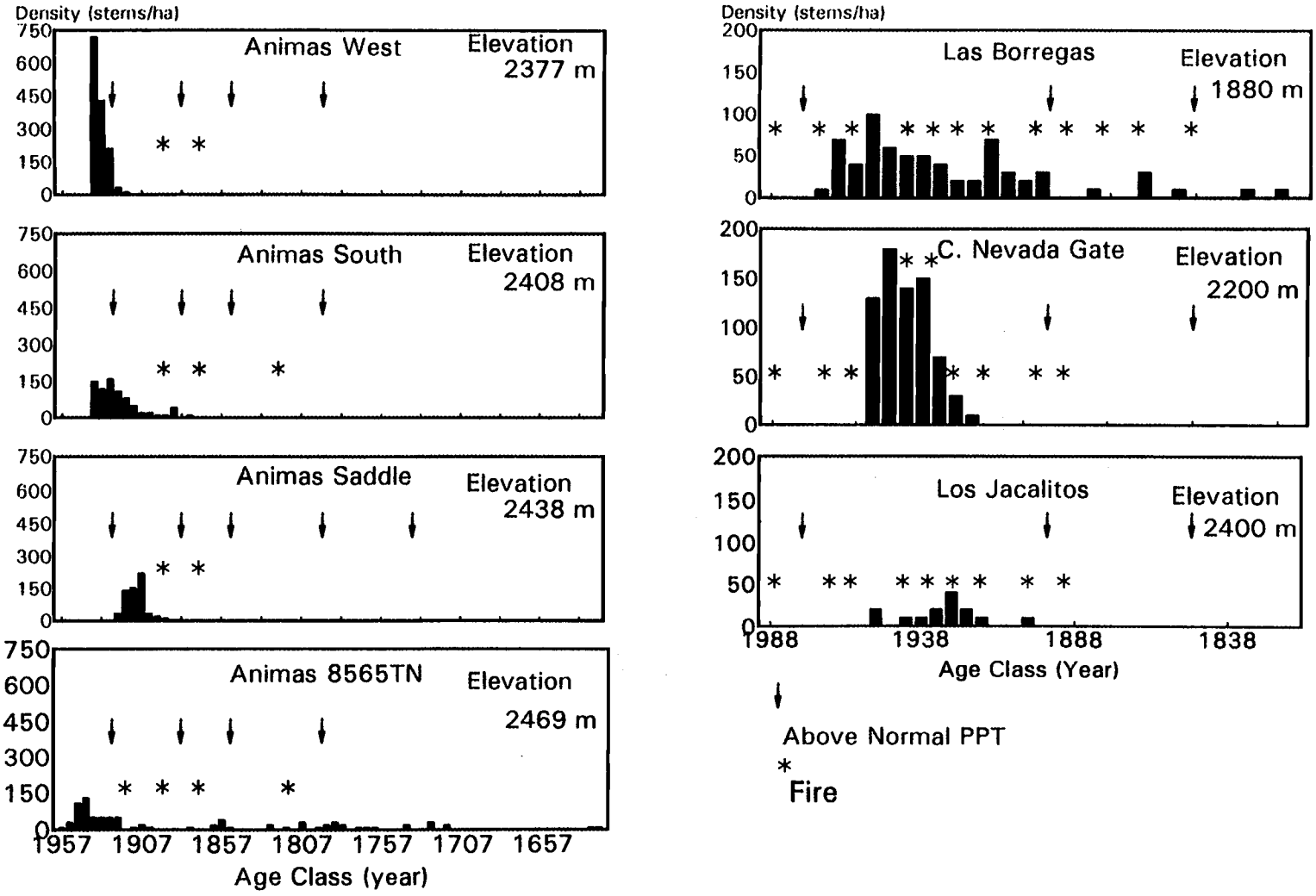

Figure 5. Relationship between fire frequency, precipitation, and establishment dates of Douglas-fir trees in the Animas Mountains, New Mexico and ponderosa pine trees in the Sierra los Ajos, Sonora.

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SOIL EROSION ESTIMATION IN THE SONOYTA RIVER WATERSHED USING THE USLE AND REMOTE SENSING AND GIS TECHNIQUES

Diego Valdez-Zamudio and D. Phillip Guertin[1]

Soil erosion is the process of detachment and subsequent removal of soil particles and small aggregates from land surfaces by wind and/or water in a specified time period (Brooks et al. 1993; Nearing et al. 1994). Soil loss due to erosion processes is a serious problem in many regions of the world.

Considering that it is impossible to estimate and monitor the soil loss effects of weather and land management practices in all ecosystems given financial and equipment commitments, and the difficulty in extrapolating beyond local study areas, an alternative approach is to develop simulation models to achieve these goals. A soil erosion simulation model mathematically describes the process of soil particle detachment, transport, and deposition on land surfaces. This kind of model could be created using a computerized database with geographically referenced systems of soil erosion to account for every parcel of land (Lull et al. 1995).

Thus, we can use erosion models: (1) to estimate current erosion effects for a given area; (2) to predict where and when erosion can occur; (3) to know about the relationship soil loss/soil erosion–crop yield/productivity; and (4) to understand erosion processes and use these experiences for assessing soil erosion planning projects, for soil conservation strategies (the rate of soil loss can be compared with what is considered acceptable), and for regulation (Nearing et al. 1994).

The USDA Soil Conservation Service created the Universal Soil Loss Equation (USLE) in 1965 to predict soil erosion. The USLE is an empirical equation designed to compute long-time average soil losses due to sheet and rill erosion across the land surface. The equation groups the interrelated physical and management parameters which influence erosion into six factors such that erosion is the product of those factors, as follows:

$$A = R\,K\,(LS)\,C\,P$$

where

- A = computed annual soil loss in tons per unit area
- R = a rainfall–runoff erosivity factor
- K = a soil erodibility factor, based on soil characteristics such as texture, structure, and permeability
- LS = a dimensionless topographic factor combining slope length, L, and slope steepness, S
- C = a dimensionless land cover–management factor
- P = an erosion control factor, based on soil conservation support practices (Brooks et al. 1993).

Some of these factors may vary spatially within a watershed and are geographic in nature: K, (LS), C, and P. The K factor is related to soil type which is normally mapped. The (LS) factor is a function of the terrain, and can be interpreted from an hypsography map. The C and P factors can be identified from remote sensing data including aerial photographs and satellite images (Potter et al. 1986). Thus, since most of the USLE's parameters can be represented geographically, it can be combined with remote sensing and GIS techniques to give better results.

Some authors have demonstrated that soil erosion models can generate accurate results when used in combination with geographic information systems (GIS) to estimate erosion in agricultural and natural areas. Lull and Tindall (1995) estimated erodibility risk values for two different watersheds (Howard Creek and Jones Meadow Creek) in western Montana using an erosion simulation model with a GIS. These watersheds differ from each other in their historical management, Jones Meadow Creek having a higher diversity of past land uses and, as a consequence, a higher risk value (2.12 vs. 1.70). They concluded that risk values obtained for these watersheds can be used as a comparative tool, on a relative basis, to evaluate watershed sensitivity to land-use changes and management.

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Other soil erosion studies have been accomplished using the USLE in a GIS environment. In 1980, Rasmussen et al. estimated the spatial and total erosion from a surface-mined area. For this study, vegetation, soils, and slope data arrays were developed for mining topsoil conditions. The form of the equation used in this analysis to determine soil loss at any cell location on the area is:

$$A_{ij} = R * K_{ij} * (LS)_{ij} * C_{ij} * P_{ij}$$

where A, R, K, (LS), C, and P represent the parameters for the USLE and ij represent the coordinates for the parameter values at a given cell in the raster grid.

This way, the output of this computer model can be displayed using different graphic formats such as computer printer maps and perspective plots.

Potter and Gilliland (1986), from the University of Nebraska at Omaha, developed a GIS called RGISM (Raster Geographic Information System for Mapping) to estimate soil loss rates in the Elkhorn River Basin, Nebraska, utilizing the USLE. The system accepts digitally mapped information on soil type, topography, and land use; it calculates slope and slope length; and it relates these characteristics to soils and land-use parameters in order to produce three-dimensional maps of runoff potential, sediment pollution potential, and fecal coliform pollution potential. After calibrating the system and analyzing the results, they concluded that the system has the potential to improve the researcher's ability to allocate mitigation resources efficiently.

Two years later, Oslin et al. (1988) from Halff Associates created STREAMS (Soil, Transport, Rainfall, Erosion, and Mapping System). This program facilitates the transfer of data between computer systems for the purpose of flood plain and watershed management. Soil type and erodibility, topography, slope, rainfall frequency and duration, land use, and vegetation cover are used in conjunction with the universal soil loss equation to compute average annual soil loss and soil erosion classes for a given watershed. Landsat TM and SPOT satellite imagery are used to derive impervious cover information and to determine changes in land use through time. Output from STREAMS maps erodible soil classification, stream erosion and deposition, and flood plain delineations under various rainfall conditions.

By using remote sensing and GIS techniques in conjunction with the USLE, this study illustrates that the Sonoyta River watershed in northwest Sonora, Mexico, may have increased soil erosion in the last two decades due to human activities, basically farming and grazing practices.

Objective

The principal objective of this study was to estimate and compare the erosion rates that occurred in 1972 and 1992 in response to land cover and land use changes in a portion of the Sonoyta River watershed, in northwest Sonora, Mexico. The objective was achieved using the USLE linked to a GIS.

Methods

The 408,381 acres studied are located in the northwestern part of Sonora, Mexico, just south of the Organ Pipe Cactus National Monument, Arizona (Figure 1). Its climate is dry with a mean annual temperature of 21.4°C, a maximum temperature of 45.1°C, and a minimum temperature of -3.3°C. The mean annual precipitation is 209.7 mm (SARH 1994).

Soil erosion processes are very closely related to land cover and land use occurring in the ecosystems. Changes in land features will be reflected in the soil loss rates for a given area. So, change detection in land cover and land use was determined for the study area following the procedure used by Valdez-Zamudio (1994). Two Landsat MSS images were processed with the remote sensing program ERDAS version 7.5 (ERDAS, Inc.), and the IDRISI for windows GIS program version 1.0 developed by Clark University, Massachusetts. The scenes used in this study were acquired from the EROS data center of the U.S. Geological Survey in Sioux Falls, SD. Those scenes are dated from September of 1972 and October of 1992.

After geometric and atmospheric correction, the 1972 and 1992 images were geographically registered. The identical geographic coordinates of the two images allow overlay operations to compare geographic changes between the two dates of the images.

To define erosion rates in the two images, it was necessary to identify and classify the objects present on those images. The criteria used to distinguish these land cover classes were based principally on topography (slope), reflectance values, the association to other features on the image such as streams, and straight borders on irrigated fields. After two field trips to the site and revising the ancillary data to obtain ground truth data, it was possible to identify training areas for a supervised classification.

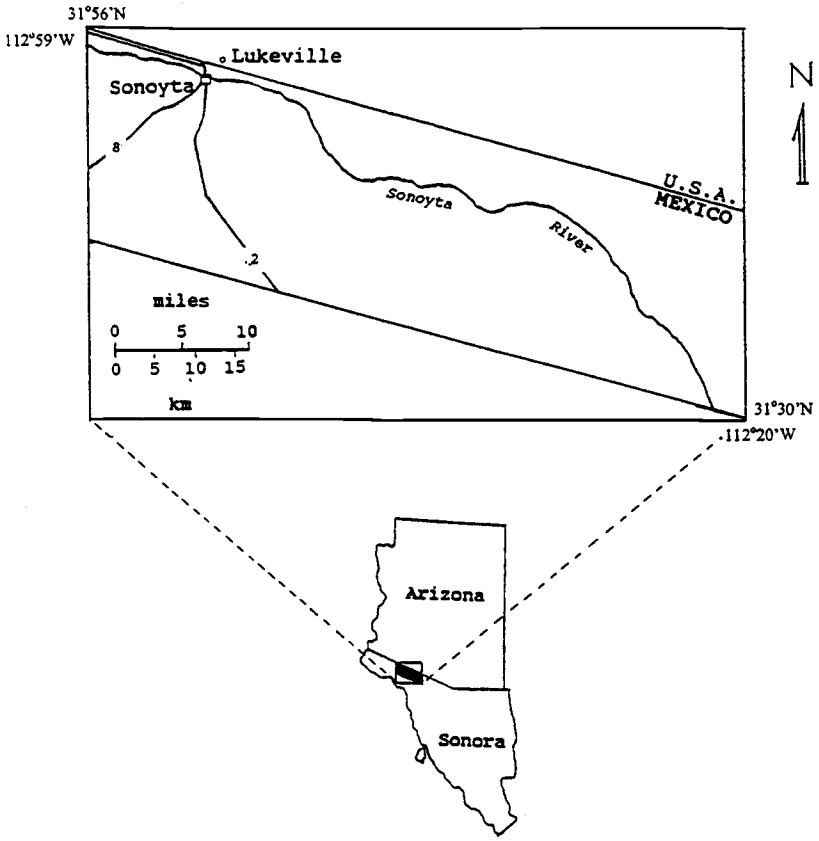

Figure 1. Extent of the study area.

When the land-use class maps were created for 1972 and 1992, the next step was to produce corresponding maps for the USLE parameters. Thus, a map layer for the R factor was created using the IDRISI program, based on the average annual values of R calculated for the state of Arizona (USDA, Soil Conservation Service 1976). The R value determined for the study area was 60.

The K or soil erodibility factor map was produced primarily by digitizing five soil-class maps acquired for the study area using the ARC/INFO program. After the digitization was performed, the vector map was then introduced into the IDRISI program to effectively grid the entire study area into a rectangular array of cells. All the cells are of the same size. Given the chemical and physical characteristics of the soils, such as texture, organic matter content, structure, and permeability, the corresponding K values were determined using a special nomograph (soil 1 = 0.20; soil 2 = 0.40; soil 3 = 0.17; soil 4 = 0.34; soil 5 = 0.40). Finally, using the IDRISI program, those values were assigned to each soil type.

In order to create a map layer representing the LS factor, it was necessary to create a slope map using the ARC/INFO program. Thus, a contour-based DEM map was produced by digitizing contour lines from a 1:50,000-scale, 20-m contour interval topographic map. Then, the slope map layer was produced using the IDRISI program. In this study we assumed that the L factor is equal to 1.0. Finally, still inside the IDRISI environment, the S factor map was obtained applying the formula:

$$S \text{ factor} = (0.43 + 0.3\,s + 0.042\,s^2) / 6.613$$

where s represents the slope gradient values (percent) in the slope map (Brooks et al. 1993).

The potential for erosion without the effect of vegetation or conservation practices was obtained by modifying the USLE to:

$$EI = R * K * L * S$$

where EI = erosion potential index (tons per acre).

The estimation of the erosion potential index resulted by overlaying (multiplying) the maps created for the USLE parameters (R, K, and S factors) using the IDRISI program. Since the EI map does not consider vegetation or conservation practices, it can be used indistinctly to compute soil loss for 1972 or 1992.

The C factor maps for 1972 and 1992 were produced by the assignation of different values to the land-cover classes according to their particular management. These values were previously calculated for Arizona by the Soil Conservation Service at Phoenix. The corresponding C values for the different land-cover classes were: croplands = 0.31, vegetation in the bajadas = 0.18, vegetation in the plains = 0.18, riparian areas = 0.01, mesquite woodlands = 0.28, and bare soil areas = 0.45. These maps were created using the IDRISI program.

As with the C factor maps, the P factor maps resulted by assigning values to the land-cover classes on 1972 and 1992 images. These values were also estimated for Arizona by the Soil Conservation Service at Phoenix. All natural areas were given a P value of 1.0, while croplands received a value of 0.37. The resulting maps were created using the IDRISI program.

The soil loss rates for 1972 and 1992 were estimated by overlaying (multiplying) the EI, C, and P factor maps by each other:

$$A = EI * C * P$$

This way, the predicted annual erosion potential (tons per acre) is stored in an output map file which can be analyzed and manipulated for achieving different studies of the area.

Results and Discussion

Five different land-cover classes were created from the image classification for 1972: croplands, bajadas with vegetation, plains with vegetation, riparian vegetation, and bare soil. Due to the amount of rain above the average between 1972 and 1992 (Figure 2), a sixth class (mesquite woodland) resulted for the 1992 image classification.

We can see in Table 1 that croplands, riparian areas, and mesquite woodlands increased in area between 1972 and 1992, while the other classes, those with negative change values, decreased in area. There were two possible factors that caused land-cover changes (positive and negative) in the study area. First, because desert vegetation is soon modified by climatic changes and considering that precipitation in the study area is the only factor of the environment that can cause a rapid change on vegetation (Cloudsley 1977), it is presumed that the amount of rain that accumulated between 1972 and 1992 increased the plant population densities, resulting in an expansion of riparian areas, the formation of mesquite woodland, and the reduction of areas with bare soil. Second, the deforestation practices executed to increase the agricultural activities in the area produced changes in land cover classes, increasing the croplands and decreasing the other classes that contributed to an increase in this class (Figures 3 and 4). These anthropogenic changes occurred as a result of Mexican government policies to generate jobs and improve the regional economy of Sonoyta by financing clearing of natural areas for agriculture and developing new wells to irrigate those new farmlands.

Table 2 shows the resulting values in soil erosion for each category in 1972 and 1992. We can

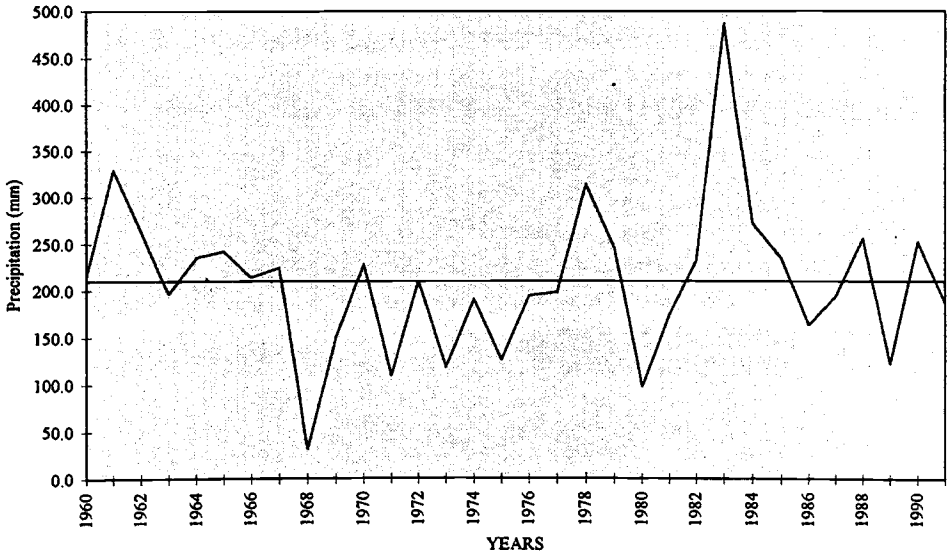

Figure 2. Historic precipitation data for study area (SARH 1994).

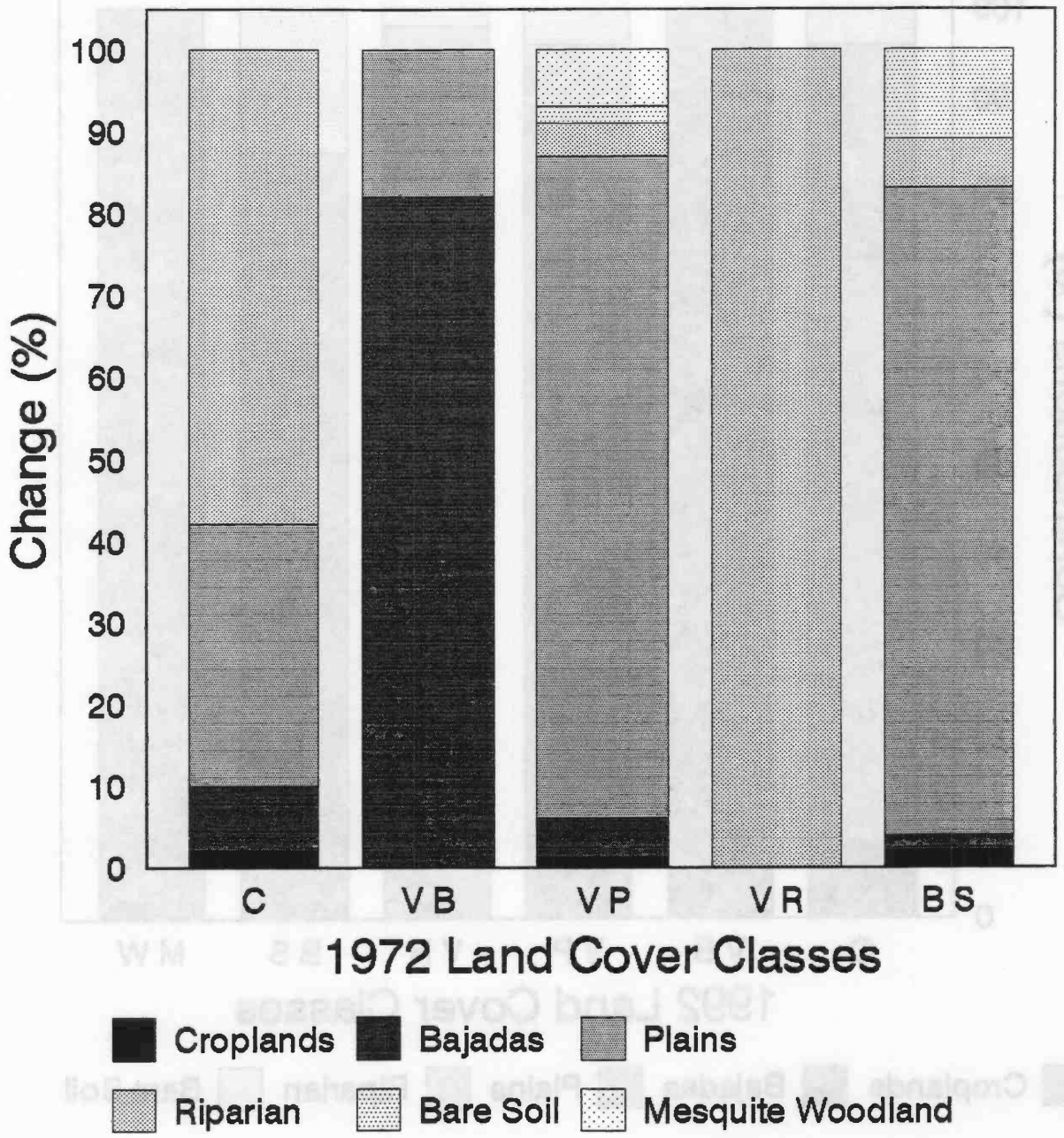

Figure 3. Different percentages of change from each land-cover class to other classes between 1972 and 1992. C = croplands, VB = vegetation in the bajadas, VP = vegetation in the plains, VR = riparian areas, BS = bare soil.

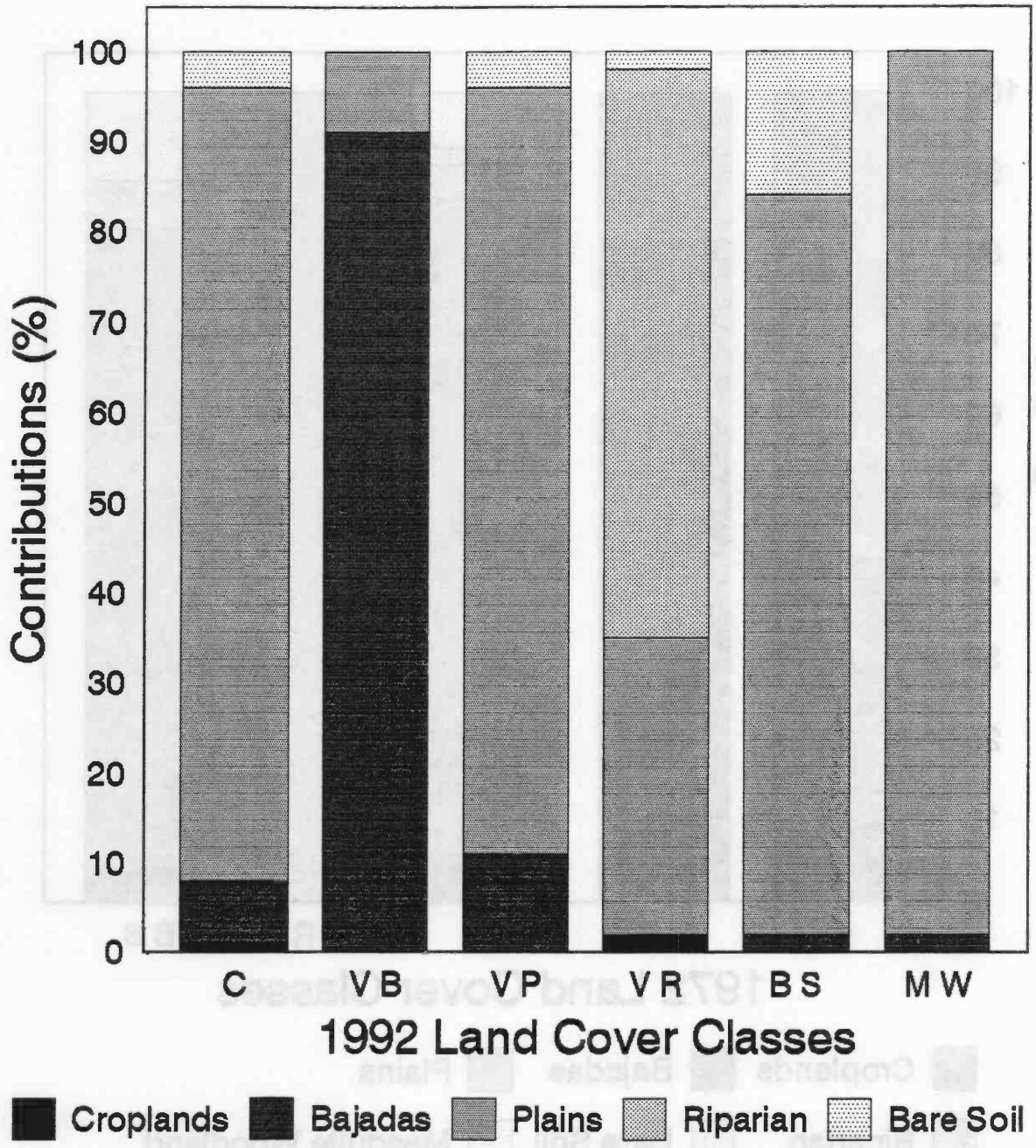

Figure 4. Percentages contribution of classes to change in individual class area between 1972 and 1992. C = croplands, VB = vegetation in the bajadas, VP = vegetation in the plains, VR = riparian areas, BS = bare soil, and MW = mesquite woodland.

see that in 1992, with the exception of riparian areas, all the land-cover classes reveal higher soil erosion rates in comparison to the same categories in 1972. In particular, croplands exhibit these erosion rates due to the tillage activities, and the other categories exhibit higher soil loss rates as a result of the introduction of cattle, sheep, and goats for grazing in densities that exceed the carrying capacity of the ecosystems.

Summary and Conclusions

This study illustrates the potential of remote sensing and GIS for making regional environmental assessments. Differences in soil erosion potential rates between 1972 and 1992 define an evident tendency of soil deterioration in the study area. A soil conservation plan needs to be implemented in this area to control the erosion processes and improve the actual conditions in the ecosystems.

Table 1. Area and percentage estimated for each category on 1972 and 1992 satellite images and percentage of change between 1972 and 1992.

Category Name	1972 Area (acres)	1972 Percent	1992 Area (acres)	1992 Percent	Percent Change 1972–1992
Crops	842.54	0.21	3567.47	0.87	323.42
Bajadas	139724.32	34.21	125793.75	30.80	-9.97
Plains	239264.75	58.59	225147.31	55.13	-5.90
Riparian	18995.18	4.65	30241.53	7.41	59.21
Bare Soil	9553.97	2.34	6549.23	1.60	-31.45
Mesquite woodland	NA	NA	17081.47	4.18	100.00
Total	408380.76	100.00	408380.76	100.00	

Table 2. Minimum, maximum, and mean erosion rates for each category in the study area. Total soil loss per category is shown in the ERC column.

Category Name	Area (acres)	Mn E R (ton/ac-yr)	Mx E R (ton/ac-yr)	M E R (ton/ac-yr)	E R C (ton/yr)
1972					
Crops	842.54	0.011907	0.504316	0.171944	144.87
Bajadas	139724.32	0.011907	1.654704	0.479293	66968.89
Plains	239264.75	0.014009	2.068381	0.479386	114700.17
Riparian	18995.18	0.007004	0.665213	0.261378	4964.92
Bare Soil	9553.97	0.011907	0.068381	0.623140	5953.46
Total	408380.76				192732.31
1992					
Crops	3567.47	0.136579	0.246346	0.189637	676.52
Bajadas	125793.75	0.126079	1.654704	0.588431	74020.94
Plains	225147.31	0.214334	0.973356	0.541421	121899.48
Riparian	30241.53	0.011907	0.054075	0.026666	806.42
Bare Soil	6549.23	0.535835	2.433389	1.308783	8571.53
Mesquite woodland	17081.47	0.333409	1.286992	0.710310	12133.14
Total	408380.76				218108.03

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A Practical Method for Estimating the Capacity of Stockwatering Ponds (Stockponds)

Don W. Young[1]

In 1979, the Arizona State Land Department (ASLD) was faced with a crucial problem. Literally thousands of water rights claims belonging to the state had been neither located nor filed upon in the pending adjudication of the San Pedro River watershed. At that time there were only four hydrologist/water resource positions within the hydrology section of the ASLD. It became obvious that a methodology was needed whereby a survey of water rights claims could be made as expeditiously as possible, while retaining an acceptable level of accuracy in quantifying those claims.

Measurement methods available at the time included (1) plane table survey, (2) instrument survey, (3) the SCS method, and (4) "guesstimate." Both plane table and instrument survey methods require two persons to perform—a luxury not afforded to the agency.

The SCS (Soil Conservation Service) method utilizes pond circumference for estimating the volume of small ponds. This method effectively treats all ponds as though they are of circular configuration and uniform depth, that is, shaped like a plate with vertical sides. Field experience has shown this not to be a reasonable assumption, and utilizing this method often grossly overestimates the capacity of stockponds.

Historically, many ranchers have simply guesstimated the capacity of their stockponds, with predictable results—volumes often varied by orders of magnitude. Primarily for this reason, ASLD decided to field measure all stockponds located on trust lands.

Each method of measurement was considered and rejected for one or more of the reasons alluded to above. Based on the experience of its field personnel, ASLD formulated a new method for estimating the volume of stockponds.

[1]Arizona Office of the Attorney General, Water Rights Adjudication Team, Phoenix, AZ.

Development of the ASLD Method

ASLD staff devised a method for estimating the capacity of stockponds, or Estimated Maximum Volume [EMV], which employs the geometric surface shape and three-dimensional configuration of ponds found in the field. Shapes and dimensional relationships used to develop this method were derived from the experience of field personnel who had visited and measured numerous stockwatering ponds, as well as actual field measurements.

Phase 1: Field observations led to the recognition of nine basic geometric surface shapes: [1] square, [2] rectangular, [3] triangular, [4] trapezoidal, [5] circular, [6] semicircular, [7] elliptical, [8] semielliptical, and [9] ellipsoidal (egg-shaped). The majority of stockponds conform to one, or a combination, of these surface shape categories (Figure 1).

Phase 2: The second phase of this project involved representation of the bottom configuration of the stockponds to develop their three-dimensional contours. Sampling of bottom configuration was accomplished through the use of a casting reel and fishing pole with a bobber modified to hold a clip through which the line would pass. An Alka-Seltzer™ tablet was inserted in the clip to keep it open, allowing the line to pass freely through the clip until the sinker reached the bottom of the pond. When the Alka-Seltzer™ tablet dissolved, the clip closed on the line. The line was then reeled in, and the length of line from bobber to sinker was measured, thereby indicating the depth of the stockpond at the point measured.

Numerous stockponds with surface configurations shown in Figure 1 were gridded and measured to develop a sample representative of bottom configuration for each shape. Other stockponds were instrument-surveyed when empty. Thus, the three-dimensional contours began to take shape, as shown by the example in Figure 2.

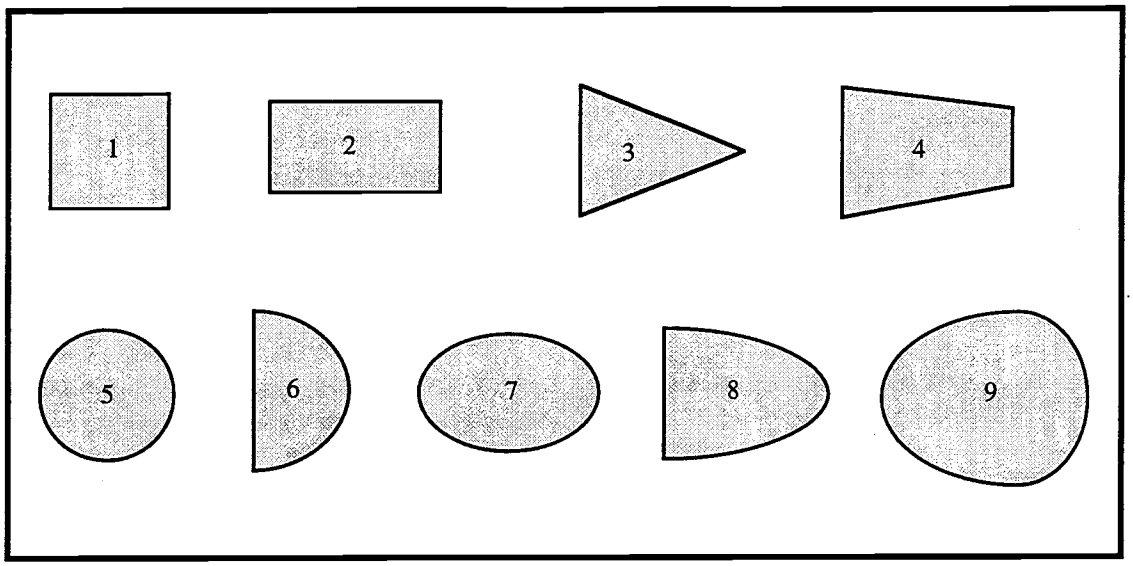

Figure 1. Basic surface configurations of stockponds identified in the field.

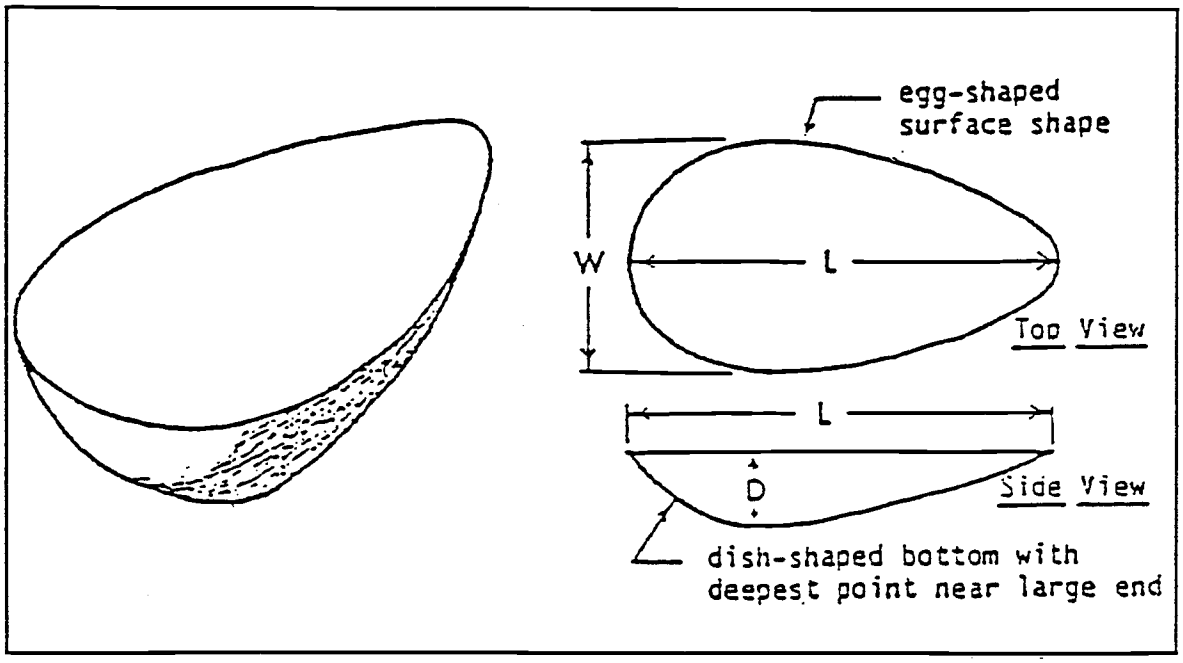

Figure 2. Three-dimensional representation of an ellipsoidal stockpond.

The next phase of the project was the most tedious. Each three-dimensional configuration had to be expressed mathematically. Geometric shapes within geometric shapes were drawn with angles and slopes, with relationships to each other carefully noted. The researchers then constructed complex mathematical formulae to express the total geometric design of each representative stockpond configuration (see Figure 3).

Next, an exact volume (V_e) formula was developed for the ellipsoidal configuration using the following assumptions:

- Volume I is one-half the volume of the ellipsoid illustrated by CN-24 (see Appendix).
- Volume II is a cylinder with an end area that is a section of a parabola. (Using a parabolic segment rather than an elliptical segment greatly simplifies the resulting equation and incurs negligible error.)
- Volume III is bounded in the xz-plane by a semiellipse having semiaxes of W/2 and 3D, and in the yz-plane by a parabola having a maximum depth D.

$$V_e = \frac{WDL(\pi+4)}{(12)(43,560)} - \frac{WD^2(7\pi+128)}{(64)(43,560)}$$

This was further simplified to an approximated volume formula:

$$V_a = 13LWD$$

and the comparative error between V_e and V_a calculated by:

$$\text{Percent error} = 100\left[\frac{0.952}{(1-3.938h)} - 1\right] \quad \text{where } h = D/L$$

(Percent error ranges between -5% and +8% for ratios of $D/L \leq 0.03$. For ratios of $D/L > 0.03$, the exact formula should be used.)

Evaluation of EMV Method

Since the development of the EMV method, the state of Arizona has been interested in obtaining data comparing various methods of measuring stockwatering ponds. Several techniques have been used by various parties to the general water rights adjudications and by the Arizona Department of Water Resources, not all of which provide consistent or accurate results (Young and Cunningham 1993, 1996).

In the early 1990s, the U.S. Fish and Wildlife Service (USFWS) was considering various methods whereby it could train its personnel to investigate water rights claims associated with its

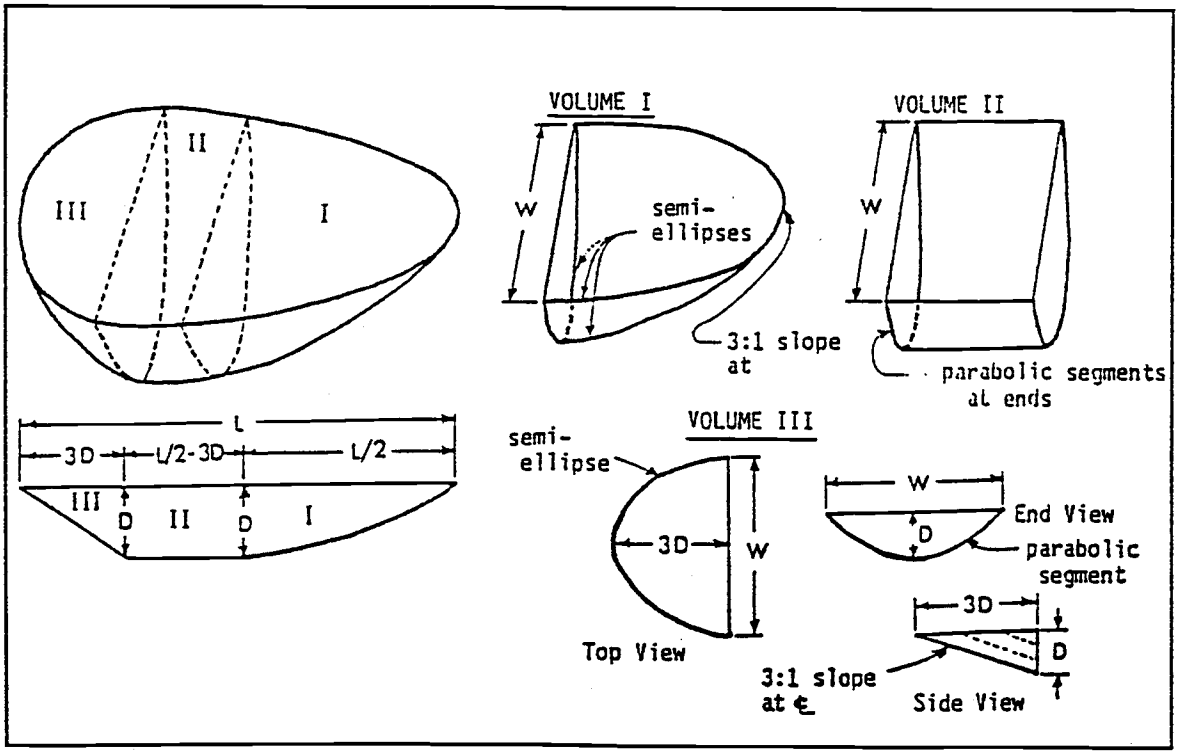

Figure 3. Geometric relationships among various segments of an ellipsoidal stockpond.

wildlife refuges in order to have the federal government's claims accurately represented, in a timely manner, in the adjudications. USFWS was interested in which methods were available, how they compared, and how (or whether) they were accepted in the context of the adjudication process (Young and Cunningham 1993, 1996).

In September 1991, the Arizona Office of the Attorney General (AG) and USFWS entered into a cooperative agreement to perform a research project on the Buenos Aires National Wildlife Refuge (BANWR), located approximately 40 miles southwest of Tucson, Arizona, to evaluate various methods of measuring stockwatering ponds (Young and Cunningham 1993, 1996).

The survey instrument used by USFWS to perform stockpond measurements is a laser-based system operating in the infrared mode to protect the eyes of the surveyor. Computer functions incorporated into the instrument permit collected data to be downloaded to a PC. Once the data are downloaded, the geodimeter surveying software permits the user to compute various survey parameters. The computer generates a contoured map of the stockpond and then computes the inside volume from the spillway elevation (top contour at the spillway) to produce the maximum volume (in acre-feet) of water that can be contained in the stockpond (Young and Cunningham 1993, 1996).

Table 1 is a summary of the data derived from the geodimeter-contoured plots and the ASLD field sheets for the stockponds surveyed. The calculated EMV of each stocktank is shown for both methods, as well as the error between the methods.

Table 1 shows the average percent error (absolute) between the two methods at just over 13 percent. In hydrologic terms this is acceptable. It must be remembered that the ASLD method utilizes capacity formulae relating to the as-built geometry and depth of stockwatering ponds. As a pond silts in, the geometry and the depth change dramatically. However, with the ASLD method the geometry would appear to remain the same. Of course, all instrument survey methods measure the existing geometry and depth of a stockpond.

In addition, the average difference in volumes of all stockponds surveyed is only 0.45 af—an insignificant volume of water when considering the potential impact of all stockponds upon a hydrologic system. Other studies indicate that retention volumes associated with stockwatering ponds have an insignificant impact on the hydrol-

Table 1. BANWR stockpond volumes measured using ASLD-EMV and GS400 methods, showing associated error (n = 17).

Stockpond Name	Volume (af)-ASLD	Volume (af)-GS400	Difference (af)	Percent Error
Ashcroft East Tank	0.47	0.34	0.13	27.7
Ashcroft West Tank	1.64	1.51	0.13	7.9
Ashcroft West Sandtrap	0.02	0.02	0	0
Halloween Tank	0.14	0.1	0.04	28.6
Herman Tank East	0.7	0.66	0.04	5.7
Herman Tank West	0.44	0.49	-0.05	-11.4
Horse Tank 1 (lower)	0.53	0.48	0.05	9.4
Horse Tank 2 (middle)	0.49	0.48	0.01	2
Horse Tank 3 (upper)	0.53	0.48	0.05	9.4
Jane's Tank	0.7	0.6	0.1	14.3
Lopez Tank	18.23	14.81	3.42	18.8
Lower Wilber Tank	0.89	0.89	0	0
McKay "A" Tank	2.48	1.99	0.5	20.2
Marijuana Tank	2.75	2.25	0.5	18.2
Marijuana Tank Sandtrap	0.82	0.68	0.15	18.3
Snake Tank	0.85	0.73	0.12	14.1
Tequila Tank	12.67	10.2	2.47	19.5
			7.66/n*	|225.5| /n*
Totals	44.35	36.71	0.45	|13.3|

ogy of a watershed (Milne and Young 1989; Young 1994a, 1994b; Young and Sejkora 1995; Young et al. 1996; Young and Lang 1996).

Conclusions and Discussion

- Utilizing the EMV method allowed ASLD field personnel (as well as ranchers and other property owners) to estimate stockpond capacities quickly and reliably. Most importantly, it required only one person to make the measurements—an asset when manpower is limited. The EMV method was utilized by ASLD in making water rights filings on the majority of its stockpond claims. Certain federal agencies and private individuals also utilized the EMV method in quantifying stockwatering ponds.
- The ASLD method produces an accurate estimation of the original as-built capacity of stockwatering ponds over a wide range of sizes and configurations—an important consideration when quantifying stockponds for water rights filing purposes. Arizona statutes permit an owner to periodically clean out stockponds that have silted in, as long as the original capacity of the tank is not increased. It is to the water right holder's benefit to file for the original capacity of the stockponds so they may be cleaned without penalty of exceeding the water right amount. This gives the ASLD method the advantage over survey methods for estimating the capacity of stockponds.
- The ASLD method is reasonably simple to perform, with a minimum of training required for field personnel.
- The ASLD method has limitations, however, with respect to impoundments that are nonconforming to standard geometric shapes and/or are very large in size (>15 af). Such unusually configured tanks would be more accurately measured by an instrument survey method.
- The ASLD method was scrutinized by the special master in the general streams adjudications in Arizona during *de minimis* hearings, and was considered by the Maricopa County, Arizona, Superior Court to be consistently accurate for its intended purpose (Superior Court Memorandum Decision 1994).
- The EMV method was originally published in 1979 and disseminated for public use by the Arizona Water Commission (predecessor of the Arizona Department of Water Resources; see appendix, "Guide For Determining Stockpond Capacity").

Acknowledgments

The author wishes to express his appreciation to Charles Graf, manager, Aquifer Protection Permitting Section, Arizona Department of Environmental Quality, who through his derivation and simplification of the capacity formulae made the development of the EMV methodology possible; also to Bradley S. McNeill, information system manager, Arizona State Parks, for his computer graphics assistance.

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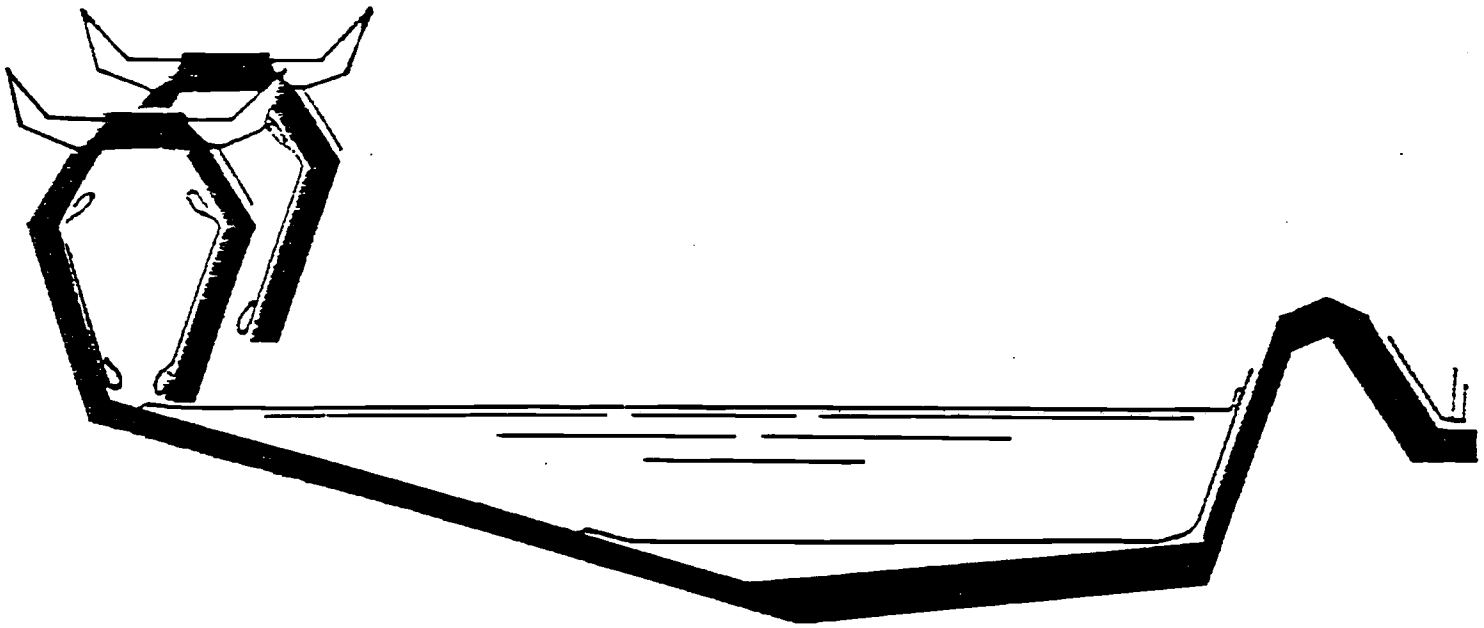
GUIDELINES FOR COMPUTING CAPACITIES OF

STOCKWATERING PONDS

AND SMALL IMPOUNDMENTS AND LAKES



ARIZONA WATER COMMISSION
REPORT NUMBER 11
1979

Appendix 1

ARIZONA DEPARTMENT OF WATER RESOURCES
GUIDE FOR DETERMINING STOCKPOND CAPACITY

INTRODUCTION

This pamphlet is designed to help answer some of the items in question 7 on the Stockpond Use Statement of Claimant Form, particularly the maximum storage volume.

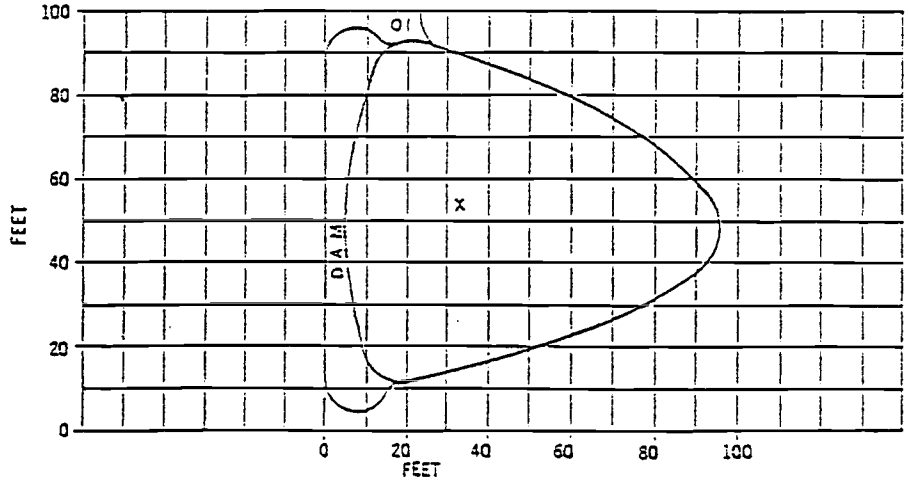
The method to compute the Maximum Storage Volume (MSV) of a stockpond consists of the following steps:

1. Sketch a top view of the stockpond—see Section I.
2. Guided by the sketch, select one of the nine basic surface shapes from Section II.
3. Select a corresponding stockpond bottom shape from the three-dimensional diagrams in Section III.
4. Compute the Maximum Storage Volume (MSV) according to the directions in Section IV.

SECTION I

On the worksheet at the end of this pamphlet, fill in the name of the stockpond in Question 1 (if not named, so indicate).

From personal observation of your stockpond, draw a sketch (using the grid in worksheet Question 2) of the top surface of the pond. The sketch should be drawn so as to depict the edge of the stockpond when *entirely full* (even if the pond is presently dry). It is helpful here if you write in foot scales along the left-hand side and bottom of the grid (see example below).



Label the dam, if one exists, by writing "DAM" at the appropriate location. The spillway should be located by placing an "O" on the sketch. If instead of a spillway the dam is equipped with a grated or ungrated overflow pipe (sometimes called "siphon" or "glory hole"), indicate this with an "X" on the drawing. If the depth of the pond is uniform throughout, place the "X" in the center of the pond and write "UNIFORM" next to it.

Now proceed to Section II.

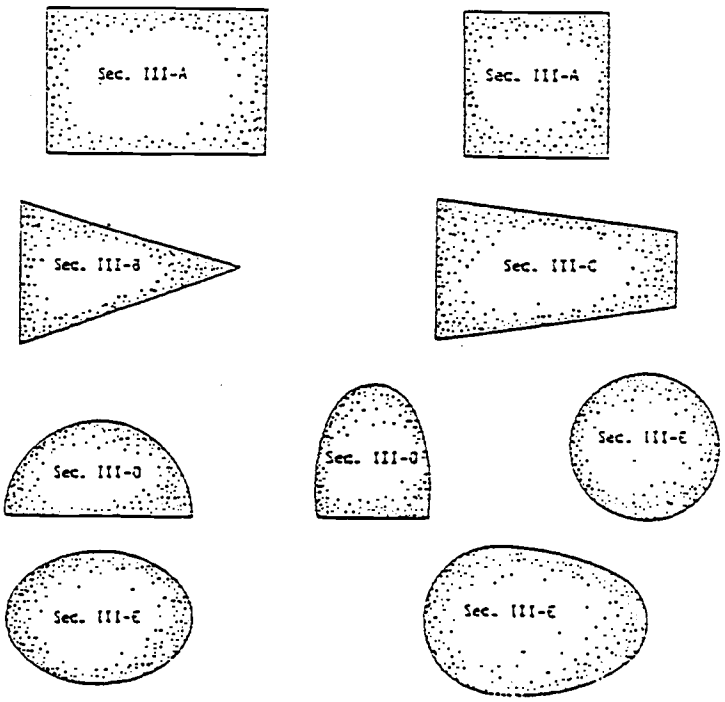
SECTION II

The following diagrams are representative of the basic surface shapes of stockponds commonly found in this region. Using the sketch you just completed, compare it with the nine (9) basic surface shapes shown below and choose the one shape which it *most closely resembles.*

Remember that you are looking at the outline of your pond, as seen from above, as it would appear if entirely filled with water. Do not worry if the surface shape of your pond does not exactly match the examples below; few will. Normally, however, after the small shoreline variations are averaged out, your pond will resemble one of the basic shapes shown. For example, the pond shown in the sketch in Section I best resembles Shape 3 – the triangle.

When you have decided which surface shape most closely approximates your pond surface, make note of the SECTION number listed for that shape, then continue to Section III.

BASIC SURFACE SHAPES OF STOCKPONDS:



SECTION III

Immediately following these paragraphs, you will find several sets of drawings that represent three-dimensional views of stockponds. Only one set contains the surface shape you picked in Section II. From that set, you will select the drawing that best conforms to the three-dimensional shape of your stockpond. If your pond is presently not dry, you will have to visualize how the pond looked when it was constructed or last dry.

Turn now to the SECTION number you noted earlier and examine the drawings, selecting the one that best approximates the shape of your stockpond. For example, the hypothetical stockpond shown in the sketch in Section I looks most like code number 16 (CN 16).

SECTION III-A

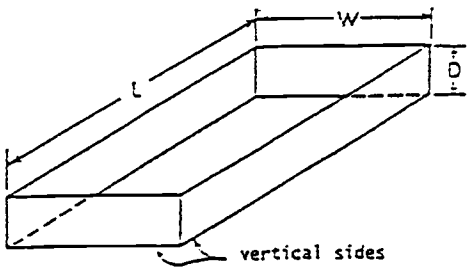
These five drawings show stockponds with a basic surface shape that is RECTANGULAR (or SQUARE). Write the code number for the drawing you select in the box at worksheet item 3, and then proceed to Section IV for further directions.

CN 1 0

Required Dimensions:

Length L
Width W
Depth D

MSV = L x W x D x 23

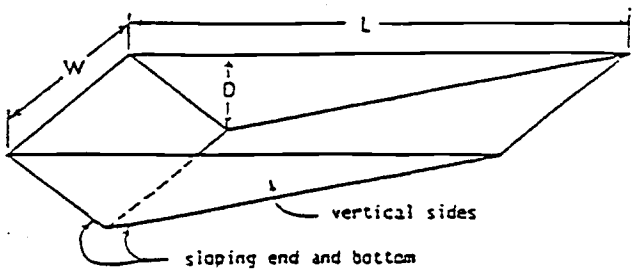


CN 1 1

Required Dimensions:

Length L
Width W
Depth D

MSV = L x W x D x 12



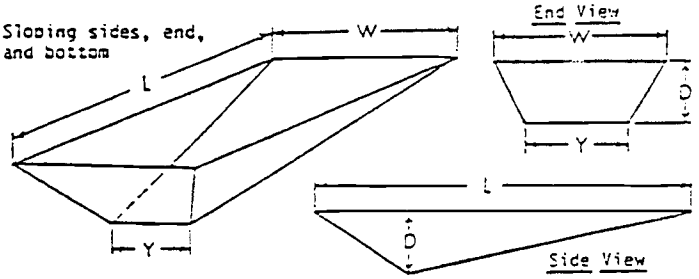
CN 1 2

Required Dimensions:

Length L
Width, top W
Width, bottom Y
Depth D

MSV = L x D x 4 x [(2xW) + Y]

Note: Sloping sides, end, and bottom

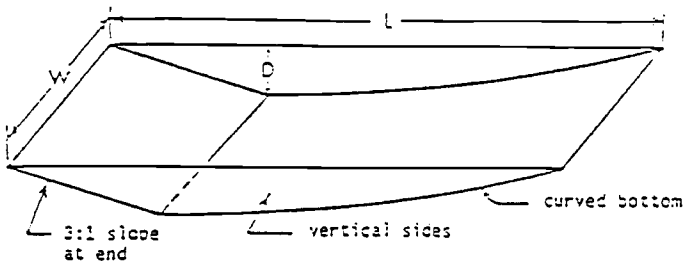


CN 1 3

Required Dimensions:

Length L
Width W
Depth D

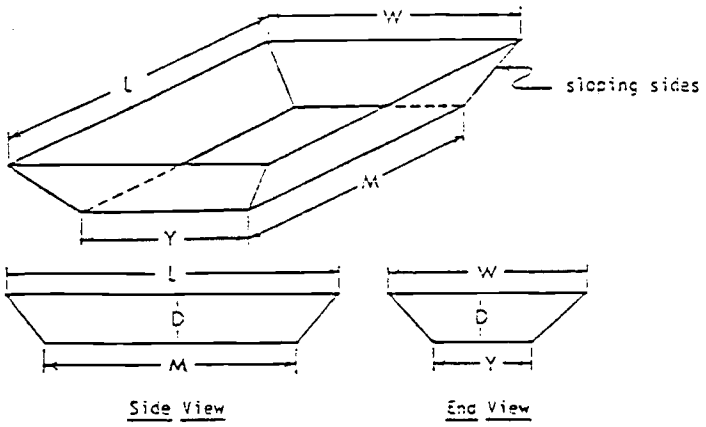
MSV = L x W x D x 15



CN| 1 | 4

Required Dimensions:

Length, top	L
Length, bottom	M
Width, top	W
Width, bottom	Y
Depth	D



MSV = 12 x D x [(LxW) + (MxY)]

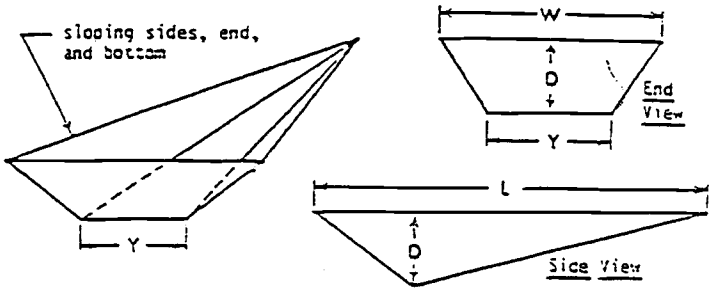
SECTION III-B

These two drawings show stockponds with a basic surface shape that is TRIANGULAR. Write the code number for the drawing you select in the box at worksheet item 3, and then proceed to Section IV for further directions.

CN| 1 | 5

Required Dimensions:

Length	L
Width, top	W
Width, bottom	Y
Depth	D

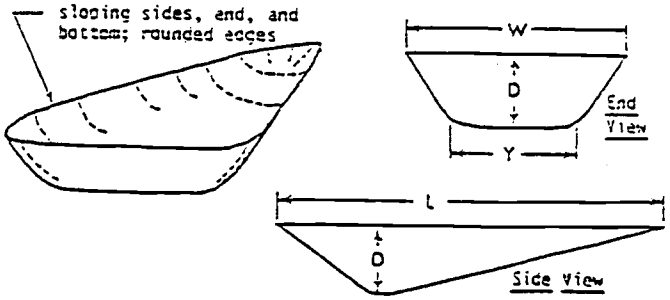


MSV = L x D x 4 x (W+Y)

CN| 1 | 6

Required Dimensions:

Length	L
Width, top	W
Width, bottom	Y
Depth	D



MSV = L x D x 4 x (W+Y)

SECTION III-C

These three drawings show stockponds with a basic surface shape that is TRAPEZOIDAL. Write the code number for the drawing you select in the box at worksheet item 3, and then proceed to Section IV for further directions.

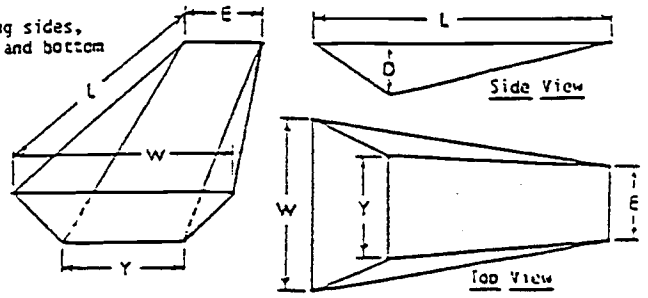
CN| 1 | 7

Note: Sloping sides, ends, and bottom

Required Dimensions:

Length	L
Width, top	W
Width, bottom	Y
Width, end	E
Depth	D

MSV| = L x D x 4 x (W+Y+E)

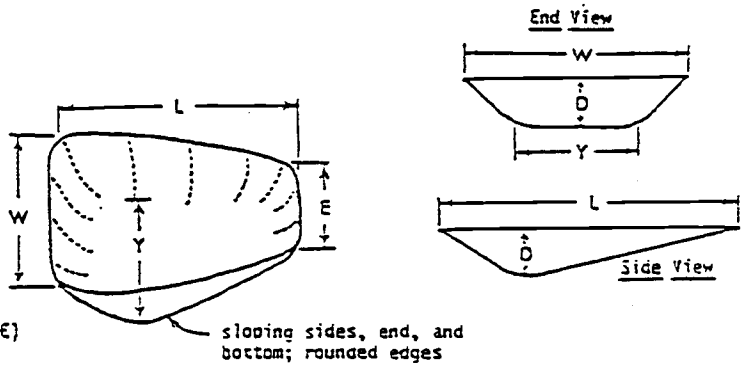


CN| 1 | 8

Required Dimensions:

Length	L
Width, top	W
Width, bottom	Y
Width, end	E
Depth	D

MSV| = L x D x 4 x (W+Y+E)

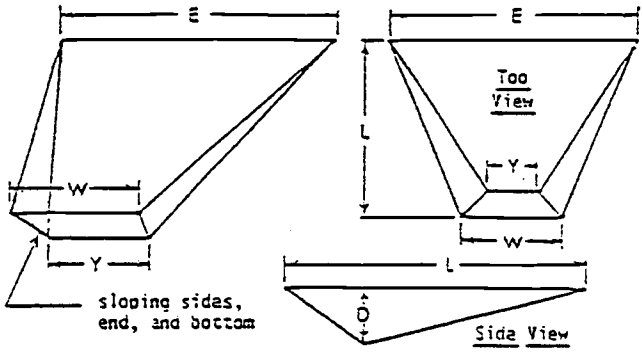


CN| 1 | 9

Required Dimensions:

Length	L
Width, top	W
Width, bottom	Y
Width, end	E
Depth	D

MSV| = L x D x 4 x (W+Y+E)



SECTION III-D

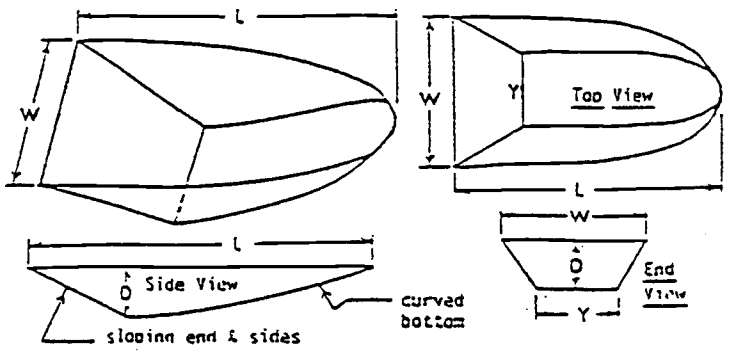
These two drawings show stockponds with a basic surface shape that is SEMI-CIRCULAR or SEMI-ELLIPTICAL. Write the code number for the drawing you select in the box at worksheet item 3, and then proceed to Section IV for further directions.

CNI 2 1 0

Required Dimensions:

Length	L
Width, top	W
Width, bottom	Y
Depth	D

MSVI = L x D x .7 x (W+Y)

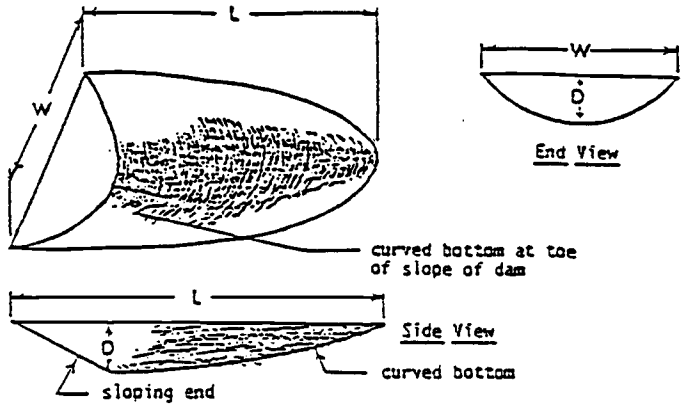


CNI 2 1 1

Required Dimensions:

Length	L
Width	W
Depth	D

MSVI = .11 x L x W x D



SECTION III-E

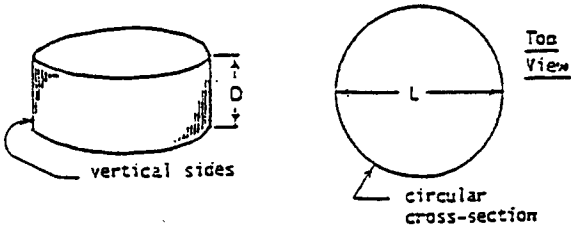
These four drawings show stockponds with a basic surface shape that is CIRCULAR, ELLIPTICAL, or EGG-SHAPED. Write the code number for the drawing you select in the box at worksheet item 3, and then proceed to Section IV for further directions.

CNI 2 | 2

Required Dimensions:

Length (Diameter) L
Depth D

MSVI = L x L x D x 18

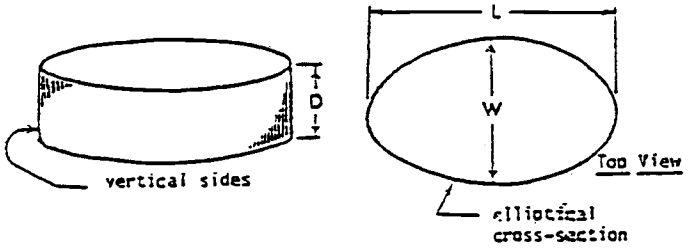


CNI 2 | 3

Required Dimensions:

Length L
Width W
Depth D

MSVI = L x W x D x 18

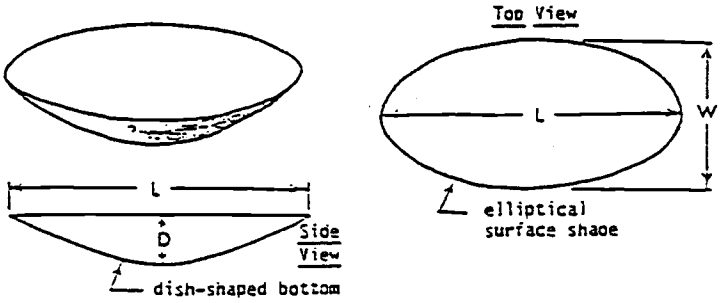


CNI 2 | 4

Required Dimensions:

Length L
Width W
Depth D

MSVI = L x W x D x 12

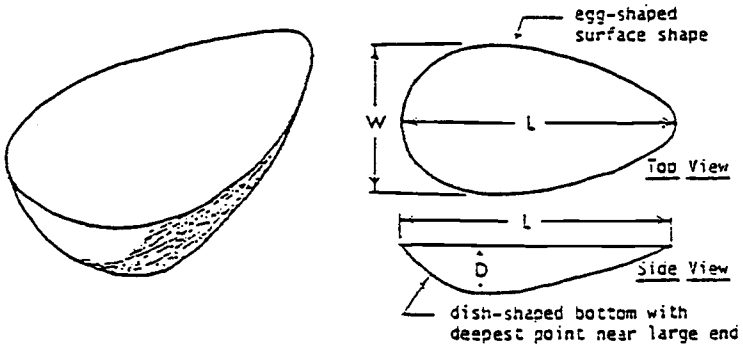


CNI 2 | 5

Required Dimensions:

Length L
Width W
Depth D

MSVI = L x W x D x 13



SECTION IV

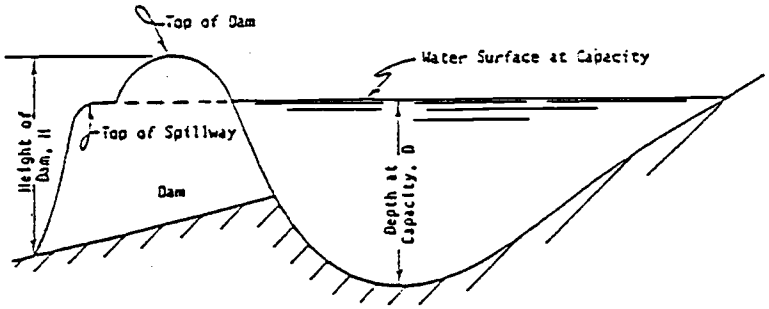
COMPUTING THE ESTIMATED MAXIMUM STORAGE VOLUME (MSV) OF YOUR STOCKPOND

A. Measurements

For each three-dimensional representation shown on the previous pages, a section headed "Required Dimensions" lists the measurements necessary to compute that stockpond's MSV. If the required measurements have not already been taken, you will have to physically obtain them by using equipment such as a tape measure, level and rod, sounding line, or your best estimate.

You should have little problem obtaining the length (L) and width (W) measurements. There may be some difficulty, however, in determining the depth (D) of your pond. Depth should be taken as the vertical distance from the top of the spillway to the deepest point in the stockpond. NOTE: THE TOP OF THE SPILLWAY IS NORMALLY A SUBSTANTIAL DISTANCE BELOW THE TOP OF THE DAM. BE SURE TO MEASURE ONLY TO THE TOP OF THE SPILLWAY (WHICH MARKS THE MAXIMUM WATER LEVEL), AND NOT TO THE TOP OF THE DAM.

If the pond is presently full, a few trials with a weighted tape measure or calibrated sounding line should locate the deepest point. If the pond is empty, the deepest point is obvious, but remember that the depth must still be measured as the vertical distance from the deepest point to the top of the spillway.



In the case where the pond is only partially filled, it is probably easiest to first locate and measure the deepest point in the pond as explained above, and then add to it the vertical distance from the water surface to the top of the spillway, thus obtaining the total depth (D) from the top of the spillway to the bottom of the pond.

The height of the dam (H) is measured vertically from the downstream toe to the top of the dam.

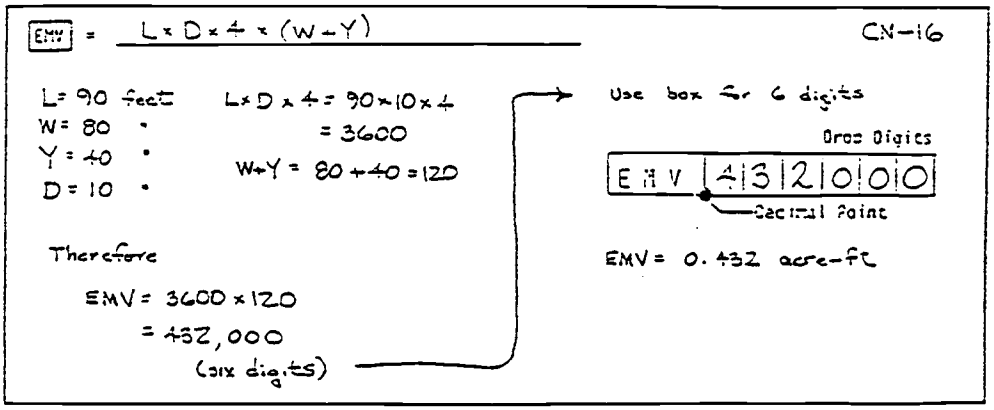
B. Calculations

The first step in calculating the maximum storage volume of your stockpond is to write down in the space provided in worksheet item 6 the appropriate MSV formula. This is obtained by referring to the three-dimensional figure you specified in worksheet item 3.

Now perform the required calculations using the measurements you recorded in worksheet item 4. The result of your calculation should be a number containing 5, 6, 7, or 8 digits.

The answer you obtain is then placed in the corresponding box in worksheet item 7; that is, an 8-digit result is written in the "8-digit answer" box, and so forth. The box automatically inserts the decimal point and drops insignificant digits. The resulting number is the volume of your stockpond in acre-feet.

An example using the stockpond sketch in Section I is worked out below:

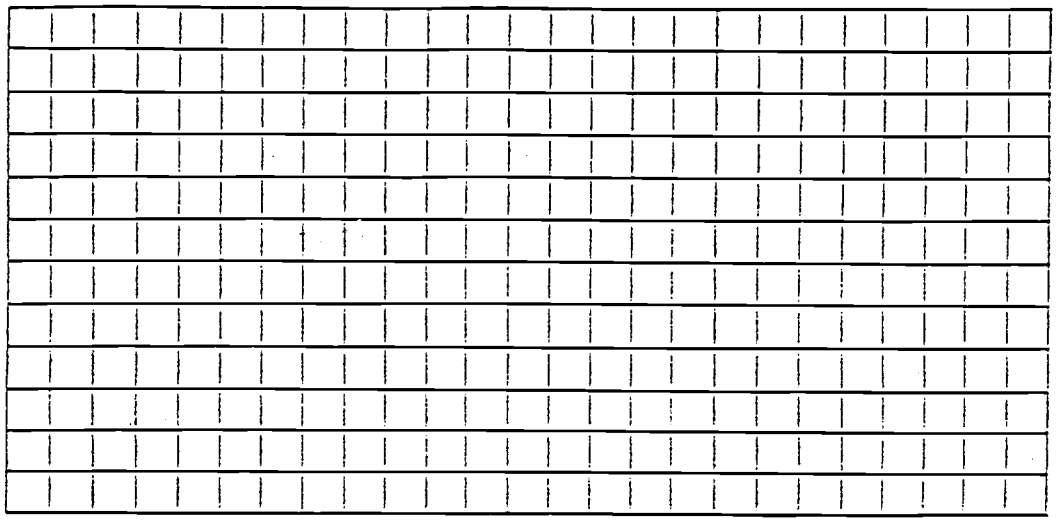


Finally, transfer to the statement of claimant form the measurements required by Question 7.

WORKSHEET

1. Name of stockpond: _____

2. Surface shape, location of dam, spillway, and deepest point:



3. Code number of selected figure: CN| | (From Section III).

4. Required dimensions for volume computation:

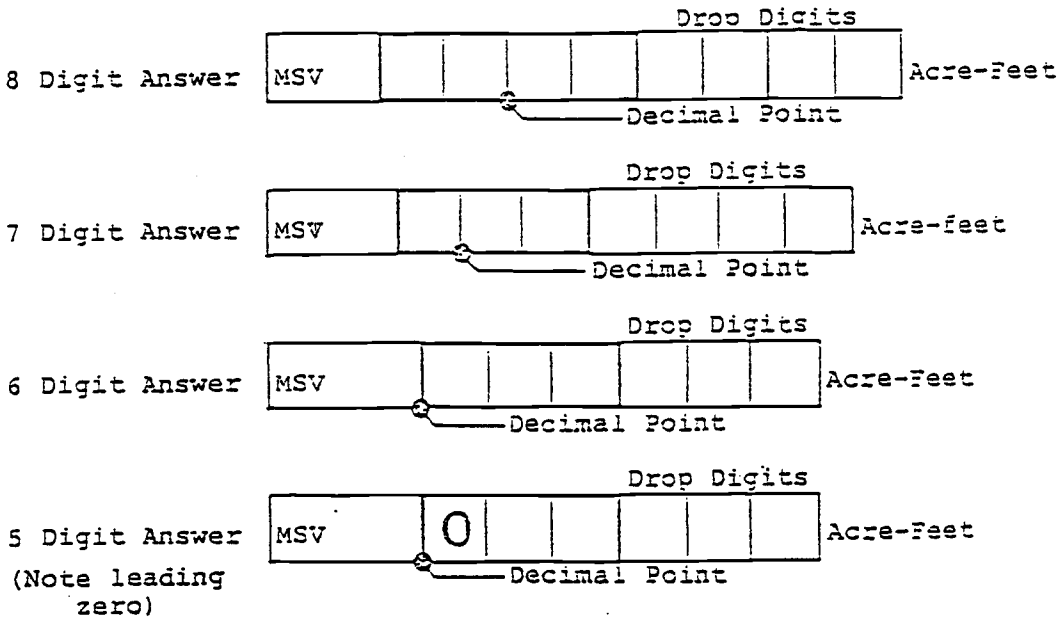
Length, Top	L	= _____	feet
Length, Bottom	M	= _____	feet
Width, Top	W	= _____	feet
Width, Bottom	Y	= _____	feet
Width, End	E	= _____	feet
Depth	D	= _____	feet
Height of Dam	H	= _____	feet

5. Calculations: Transfer the equation associated with the figure selected from Section III to the space following. Do the required calculations using the dimensions of your stockpond listed above.

WORKSHEET (continued)

MSV = _____

7. Maximum Storage Volume (MSV). Transfer the answer you calculated in the previous step to the appropriate box below.



When transferring the volume calculated above to your statement of claimant form, retain only the digits (and decimal point) located within the heavily outlined portion of the box.

EFFECTS OF HISTORIC MINING ON GROUNDWATER AND SURFACE WATER

Ulrike Rösner[1]

Ore mining developed into a booming business in the southwestern United States in the second half of the last century, with innumerable small mines opening and mining towns sprouting up almost everywhere. However, most of the smaller mines were given up at some stage before the late 1940s for a variety of quite different reasons.

A typical example for the boom and bust of ore mining in the Southwest can be found in the Cerbat Mountains, Mohave County, Arizona (Figure 1). Small mines were to be found particularly frequently in the areas east and southeast of the little town of Chloride. The miners were looking for gold and silver, and later on also for lead, zinc, and copper (Dings 1951). Some of the mines—primarily the Tennessee Mine, which was the largest producer at that time—even processed the ore on site. Today the mines are abandoned, but numerous tailings and waste rock dumps remain.

Strange-colored deposits can easily be recognized in several streambeds below the old tailings—fine sediments washed out from the tailings and dumps during heavy rains. In the light of this evident pollution and considering the widespread mining activities, the question arises: to what degree could the remnants of the deserted mines affect the groundwater and the surface-water quality in this historical mining district?

But before going into the investigation results, a brief survey of the main physicogeographical facts of the study area seems to be appropriate.

The Study Area

The Cerbat Mountains and the western adjacent Sacramento Valley Basin are part of the Basin and Range Province. The climate is arid to semi-arid with average precipitation rates of 6 to 10 inches at the western foot and 12 to 20 inches in the mountains above 4,000 feet. The stream flow in the upper reaches of the Cerbats is intermittent—flowing continually for several months—whereas the stream flow in the valley and its margins is ephemeral.

The hydrogeologic situation comes about through the difference between range and basin (Figure 2). The igneous and metamorphic rocks (granite, gneiss, shist) of the range generally do not yield water except along fractures and in weathered zones. Wells located at the foot of the Cerbat Mountains are completed in the zone of fractured/weathered rocks. The principal aquifer in the Sacramento Valley region is the older alluvium (Gillespie and Bentley 1971; ADWR 1990).

Methods

Field work was carried out in spring and fall of 1995. Twenty-seven water samples were then analyzed at McKenzie Laboratories, Phoenix, for their general chemistry and for 12 heavy metals (As, Ag, Cd, Cr, Cu, Hg, Ni, Pb, Se, Zn, Fe, Mn). Judging from background samples (WP 8, WP 11, and WP 17), the water in the Cerbat Mountain area probably met drinking water quality before mining started. Drinking water standards were therefore used as a comparative basis: first the official DWS = Domestic Water Source standards, enforceable standards published in the Arizona Administrative Code (AAC) (ADEQ 1995); second the Health-Based Guidance Levels (HBGLs) for drinking water and soil, non-enforceable levels set by the Arizona Department of Health Services (ADEQ 1992).

Results

Only some examples of the study can be discussed in this short paper (for complete data concerning the water quality see Rösner 1995).

Surface Water

The first indications of a contamination are obtained by field measurements of the electrical conductivity (EC) and the pH values.

The EC values in the Tennessee Wash (Figure 3) gradually increase on its way through the canyon, in which one abandoned mine closely follows the

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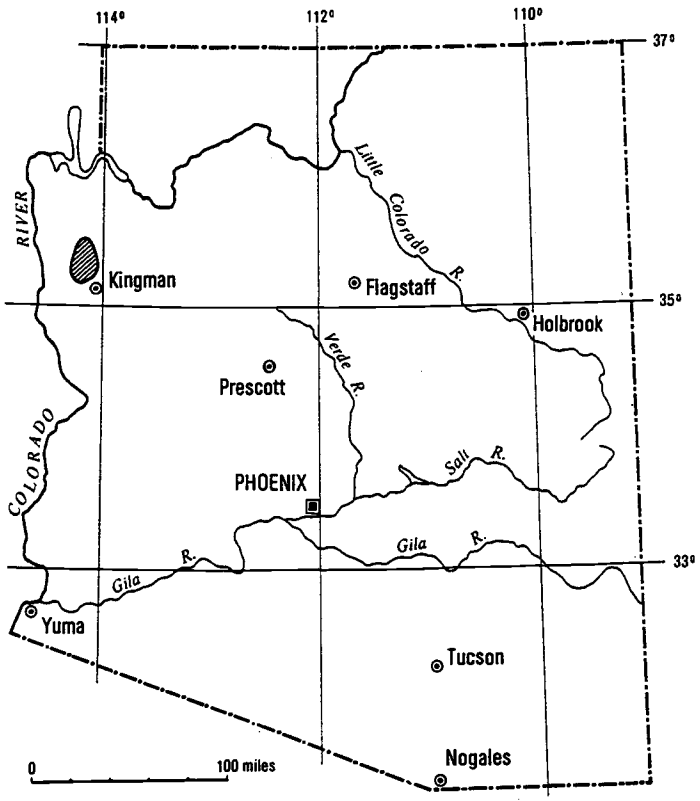

Figure 1. Map showing area of report (shaded) in Mohave County, Arizona.

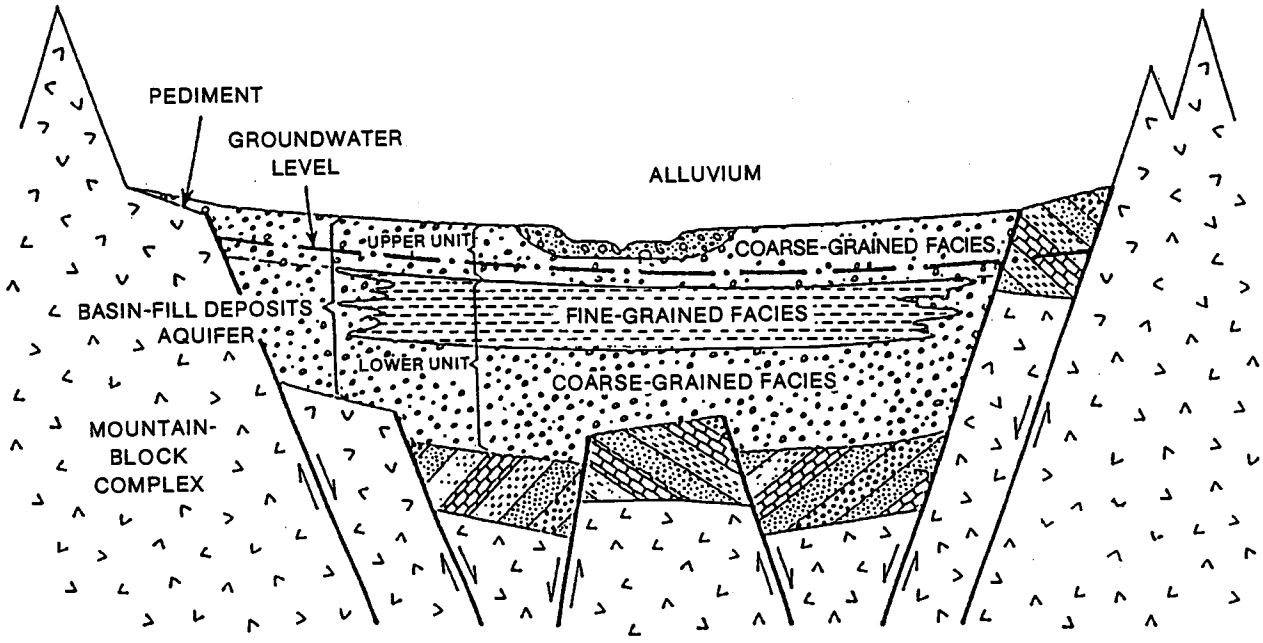

Figure 2. Basin and Range hydrogeologic section (after ADWR 1990).

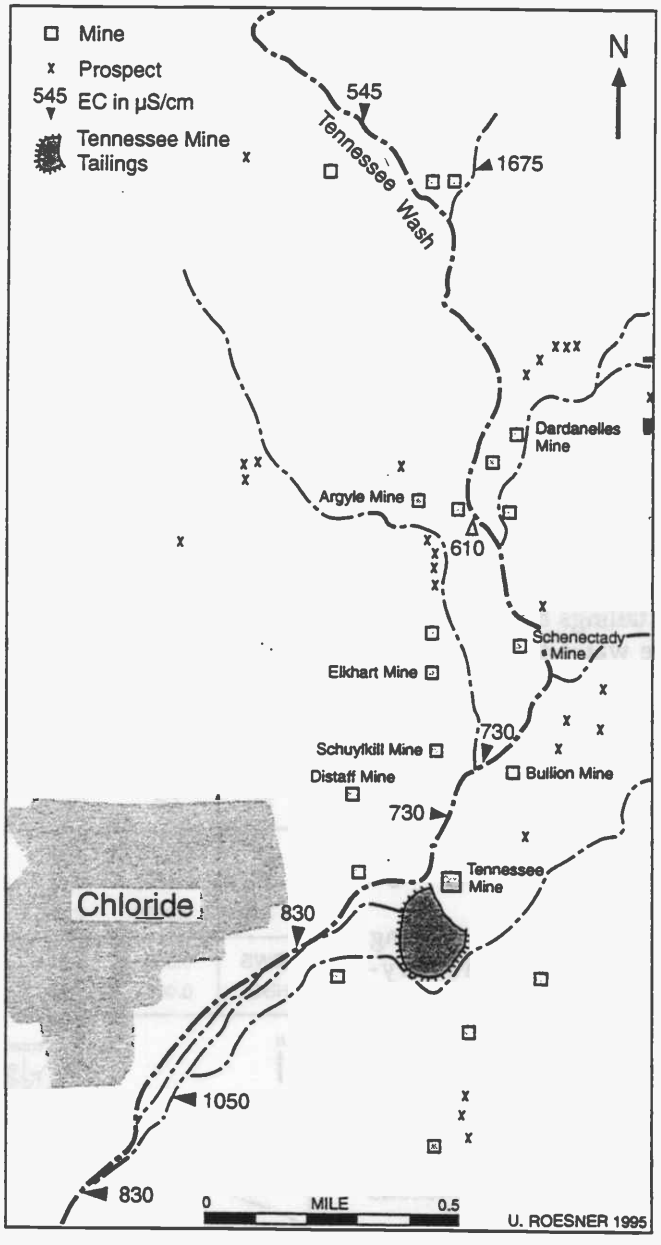

Figure 3. EC values along Tennessee Wash (March 1995).

other (starting from 545 µS/cm). The maximum of 1,050 µS/cm is reached below the confluence, with small washes flowing directly through the area of the Tennessee Mine tailings and the western adjacent mines. Further downstream the electrical conductivity decreases again to 810 µS/cm. Consequently, this steady increase of EC values indicates that chemical substances are being washed out of the tailings and are entering the surface water.

Similar tendencies are shown by the pH values (Table 1). They are considerably lower in tunnel or tailings discharges or in discharges from active mines such as the Cyprus Mineral Park Mine. Both the EC values and the pH values point to a contamination, but they do not say anything about the nature, the degree, and therefore the dangerousness of the water pollution. This information can be obtained by the heavy-metal concentrations.

Table 2 (for location of samples see Figure 4) shows heavy-metal concentrations in the surface water of the Chloride mining district east of Chloride. The samples exceed the standards for arsenic, cadmium, chromium, copper, nickel, zinc, iron, and

Table 1. Surface water pH values in the western Cerbat Mountains mining area.

Water Source	pH
Clear surface water	7.0–8.3
Discharge of old tailings	4.9–7.0
Discharge of old tunnels	3.0–6.6
Discharge from the operation site and the tailings of the Cyprus Mineral Park Mine	2.6–3.2

manganese in different combinations and proportions. Samples WP 21 and WP 5 in particular—the sampling sites from right beside and just downstream of the big Tennessee Mine tailings—show very heavy contaminations. These results prove that remarkable amounts of heavy metals are being washed out of the old tailings by heavy rains or are entering the surface water from polluted, shallow groundwater.

The surface water was also found to be polluted in canyons that were not as heavily mined as Tennessee Canyon. In the case of Eureka Canyon (Figure 5), the streambed was covered with a light-blue, soft deposit in the lower section, the color of which is most likely due to its extremely high content of copper: 1,000 mg/L.

The discharge of an old tunnel (WP 27) flowing into Eureka Wash has a particularly high heavy-metal load which pollutes the streamflow of the main wash below the confluence (WP 28 compared to WP 26). For example, the level of cadmium is 32 times higher than the DWS standard, and copper is 41 times higher (WP 27). Even after dilution with cleaner water from the upper canyon, the heavy-metal concentration still clearly exceeds appropriate levels.

Such tunnel discharges can occur several times in a single canyon (e.g. the canyon of the Golconda mining area south of the Chloride mining district). Consequently, the streamflow is repeatedly enriched by heavy metals on its way through the canyon. The same effect occurs when several tailings in a canyon line up along a wash; in contrast to that, simple waste-rock dumps have a negligible effect on the water quality.

When some of the water samples show a higher reading for certain heavy metals but do not exceed standards, one has to keep in mind the fact that the heavy rains on the days before sampling will presumably have diluted the load of pollutants.

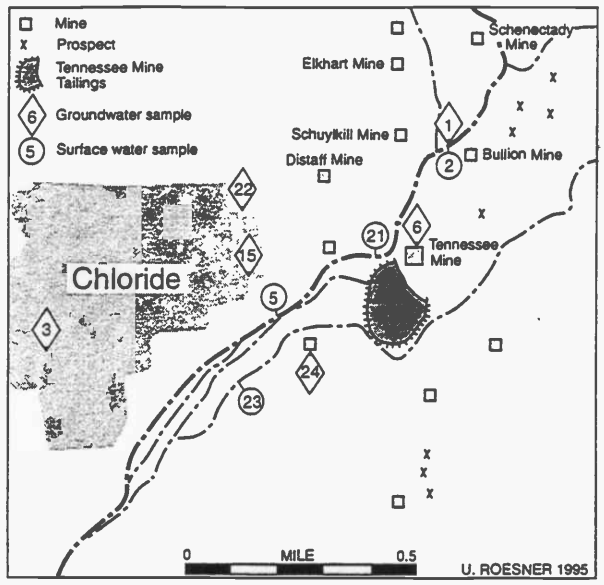

Figure 4. Location of surface water and groundwater samples in the Chloride mining district and in Chloride itself.

ID	Cd	Cu	Ni	Zn	Fe	Mn
26	0.0190	1.20		8.40		
27	0.1600	41.00	0.73	23.00	81.0	8.60
28	0.0220	2.20		8.60	2.5	0.75
DWS	0.0050	1.00	0.14	5.00	NA	NA
HBGL	0.0035	1.30	0.14	1.40	0.3*	0.70

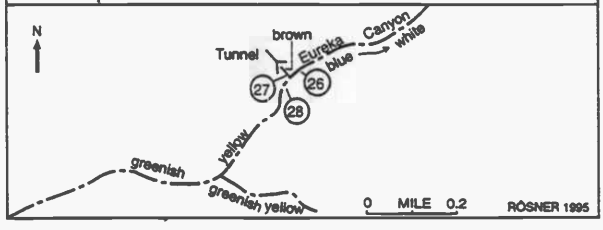

Figure 5. Heavy-metal concentrations in the surface water of Eureka Canyon exceeding DWS and HBGL standards (units are mg/l).

Groundwater

When the surface water is already more or less contaminated, the following question arises: to what extent are the pollutants from the mining areas indeed entering the groundwater?

Comparative investigations of a well outside any influence of former mining activities proved that groundwater at the western foot of the Cerbat

Table 2. Heavy-metal concentrations in the surface water of the Chloride mining district exceeding DWS and HBGL standards (units are mg/l; for location of samples see Figure 4)

Sample	As	Cd	Cr	Cu	Ni	Pb	Zn	Fe	Mn
2	0.068					0.029		9.7	
5	0.098	0.0195	1.1			0.170	5.9	4.2	
21		0.6100		1.5	0.22	0.018	200.0	5.7	15.0
23	0.080	0.0062						0.5	
DWS	0.050	0.0050	0.1	1.0	0.14	0.050	5.0	NA	NA
HBGL	0.050	0.0035	0.1	1.3	0.14	0.005	1.4	0.3*	0.7

Table 3. Heavy-metal concentrations in the groundwater of the Chloride mining district and of Chloride itself (for location of samples see Figure 4).

Sample	As	Cd	Pb	Zn	Fe	Mn
Groundwater in the Chloride mining district						
1	0.122	<0.0005	<0.005	<0.05	0.35	<0.05
6	0.999	0.0068	2.000	3.30	77.00	6.40
24	0.491	0.0189	0.650	10.00	6.40	4.80
Groundwater in Chloride						
3	0.008	<0.0005	<0.005	<0.05	0.47	0.15
15	0.050	<0.0005	<0.005	<0.05	<0.05	<0.05
22	0.011	<0.0005	<0.005	0.09	0.18	<0.05
Background sample						
17	<0.005	<0.0005	<0.005	0.38	0.96	0.38
DWS	0.050	0.0050	0.050	5.00	NA	NA
HBGL	0.050	0.0035	0.005	1.40	0.3*	0.70

Mountains is naturally highly mineralized: the EC values go as far as 4,000 µS/cm (TDS up to 2,700 mg/L). Therefore, electrical conductivity is not usable as a contamination indicator for the groundwater.

Likewise, the pH values do not admit any conclusions regarding contamination due to the old mines, because the values range in an undifferentiated manner between 6.8 and 7.9—no matter whether the wells are located in the mining area or not.

In contrast, the heavy-metal distribution reveals the following differences. In the groundwater samples from Tennessee Canyon (WP 1), the Tennessee Mine main shaft (WP 6), and the well south-southwest of the large tailings (WP 24), the permitted DWS standards are exceeded for arsenic, cadmium, lead, zinc, iron, and manganese; the arsenic and lead values of WP 6 are particularly far in excess of the DWS standards, with arsenic 20 times and lead 40 times higher than the standards (Table 3). In spite of these high heavy-metal concentrations in the Chloride mining district, the Chloride aquifer (WP 3, WP 15, WP 22) has been unaffected.

Three explanations should be considered for the lack of heavy metals in the Chloride groundwater. The first explanation would be that there is no groundwater flow from the nearby mining district east of Chloride to the town's aquifer. This seems to be the most likely explanation, because the groundwater in the proximal mountain foothill zone is found in the fractured/weathered zone of the Precambrian igneous and metamorphic rocks (ADWR 1990, p. 6). For that reason, it is possible that no uninterrupted groundwater aquifer exists;

rather, different groundwater systems, separated from each other, may be present. This interpretation is also supported by two facts. First of all, the water level in the Tennessee Mine main shaft (4,160 feet) is much higher than the groundwater table in Chloride (e.g. 3,970 feet at the Fire Department well). Second, the wells in the higher Tennessee Mine area have not run dry during hot summers, as has happened with some wells in Chloride.

As a second explanation, a dilution effect after the rainy season was taken into consideration. But additional groundwater samples, taken in September 1995 after the dry summer period, showed no difference to the results of the same wells sampled in March 1995. The dilution explanation can therefore be excluded.

A third explanation could be the influence of specific solubility and mobility of heavy metals in soil and groundwater on their transport. It could thus be possible that the heavy metals are being immobilized in the soil by chemical reactions and adsorption (see Brümmer et al. 1986; Hornburg and Brümmer 1990; Scheffer and Schachtschabel 1992; Hütter 1994). Considering the high concentrations of heavy metals in the immediate vicinity of the mines (e.g. WP 6, WP 24) and the relatively short distance to the Chloride wells (0.5 to 0.8 miles), however, that explanation appears questionable.

Summary

To return to the initial question, the effects of historic mining on surface-water and groundwater quality can be summarized briefly as follows:

1. Just downstream of old tailings and tunnel discharges, the surface water is highly affected by pollutants from the upstream historical mining remnants. The concentration of the pollutants decreases relatively soon—about a half mile to a mile below the last tailings or the last confluence with tunnel discharge. However, the distance required until recovery takes place depends on different parameters, such as the extent of the tailings, the rate of tunnel discharge, the type and degree of contamination, and the present streamflow.

2. The groundwater in the immediate surroundings of old mines which carried out ore processing on site is heavily contaminated. In contrast, the groundwater in the foot zone of the Cerbat Mountains obviously stays unaffected by any upstream contamination of historical mining activities. The most likely explanation seems to be that there is no uninterrupted groundwater flow from the nearby mining district in the fractured rock zone along the base of the Cerbat Mountains.

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AQUIFER CONTAMINATION AND SAFE DRINKING WATER: THE RECENT SANTA CRUZ COUNTY EXPERIENCE

Terry Sprouse, Dennis Cory, and Robert Varady[1]

Santa Cruz County is located in southern Arizona, north of the Mexican state of Sonora (Figure 1). The Santa Cruz River and its tributary, Nogales Wash, flow northward from Nogales, Sonora, through Nogales, Arizona, and up to Maricopa County, south of Phoenix. Over the past decade, Santa Cruz County has experienced an unusually high number of problems attributed to water quality. As a recent example, trichloroethylene (TCE) has been found in groundwater near the Potrero Wash (Woodward-Clyde Report 1995). Additionally, raw sewage, petroleum, heavy metals, DDT, parasites such as giardia and cryptosporidium, and fecal coliforms, have been found in Nogales Wash (Varady and Mack 1995). Further, elevated levels of nitrate and bacteria also have been found in wells near Nogales Wash and along wastewater lines. The Nogales International Wastewater Treatment Facility, which is in Arizona, downstream from Nogales, Sonora, does not eliminate bacteriologic contamination caused by wastewater (Varady and Mack 1995). While only an intermittent problem, these contaminants do pose a potential threat to the underlying aquifer and to the health and lives of humans and livestock that may come into contact with the water.

Following heavy rains in October 1990, volatile organic compounds (VOCs) such as tetrochloroethylene (PCE) and TCE were found to be at such high levels that the Santa Cruz board of supervisors declared a health emergency. Such seasonal rainstorms also commonly cause untreated sewage to run directly into the wash and through downtown Nogales, from Mexican sources such as outhouses, industry, septic tanks, and sewer lines. The combination of physiography and human action has resulted in other, related environmental crises. In February 1994, for example, 2,000 residents had to be evacuated from downtown Nogales, Arizona, because of the presence of potentially explosive fumes emanating from Nogales Wash, the result of petroleum products dumped into the wash in upstream Sonora (Varady, Ingram and Milich 1995).

Among the water-related public health issues causing the most concern have been the identification of 14 incidences of multiple myeloma and 26 cases of lupus in Santa Cruz County between 1989 and 1993. The expected numbers of multiple myeloma and lupus cases for a population the size of Santa Cruz County are five and four, respectively (Varady and Mack 1995). Although epidemiologists have yet to establish a causal relationship, affected residents believe these problems are caused by environmental contamination, possibly because of TCE in the drinking water (Varady and Mack 1995).

In response to the growing crises, and recognizing the lack of reliable environmental data from this region (Carter et al. 1996), the Arizona Department of Health Services announced, on November 24, 1995, the opening of the Sonora-Arizona Border Public Health Office in Nogales, Sonora. Health officials, including a team from the Center for Disease Prevention and Control, are studying the link between environmental pollutants and cross-border health problems. Data related to incidences and causes of diseases will be shared by the two countries at the new office, which is organized in part by the Governors Binational Border Health and Environment Task Force (Tucson Citizen, Nov. 25, 1995).

Geography of Santa Cruz County

Low rugged mountain ranges and deep alluvial valleys characterize southern Santa Cruz County. The elevation of Nogales is about 1200 meters and it lies in a narrow valley near the Santa Cruz River. Low hills surround Nogales, but beyond the Santa Cruz River to the east and north lie higher mountains (Udall Center 1993).

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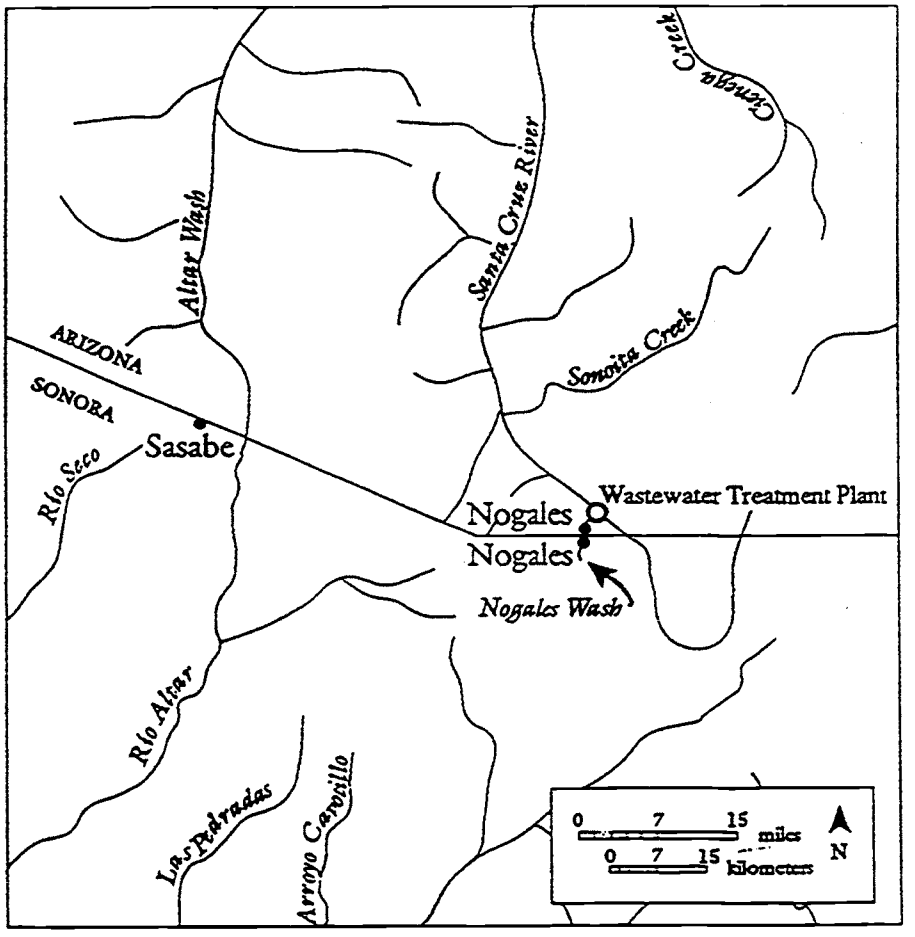

Figure 1. The Santa Cruz River basin and surrounding area (from Tellman et al. 1991).

The major drinking-water well sites for both Nogales and most of the population of Santa Cruz County lie along the Santa Cruz River. The Santa Cruz River originates in the San Rafael Valley, about 20 miles east of Nogales, flows southward into Mexico, and then turns east and reenters the United States about 5 miles east of Nogales. Typical water flows returning from Mexico range from less than 1 cubic feet per second (cfs) in summer to 30 cfs in winter. The stream channel from 2 miles north of the U.S.–Mexico border to Guevavi Ranch, located 6 miles to the north, is usually dry (Lawson 1995).

Nogales Wash originates in Mexico and flows northward through Nogales until it connects with Potrero Creek and flows into the Santa Cruz River. Nogales Wash normally would be dry if not for the discharge of water from leaky wastewater and water-supply lines in Sonora. Just downstream of the confluence of the Santa Cruz River with Nogales Wash lies the International Waste Water Treatment Plant, which treats 14–15 million gallons per day of wastewater from both Nogales, Arizona and Nogales, Sonora (Udall Center 1993). The treated effluent from the plant is dumped into the normally dry Santa Cruz River. The rejuvenated river flows above ground through Tubac, until it disappears underground before reaching the Pima County line. The dry river bed continues north, heading to Phoenix, where it connects with the Salt River.

Population growth in Santa Cruz County has been dramatic over the past four decades, quadrupling between 1950 and 1987—increasing by 530 percent in Nogales alone. Between 1975 and 1995 the population in Santa Cruz County rose from 17,300 to 32,400. The majority of this growth has taken place in Nogales, the county seat and largest city. Projections indicate that the populations will rise to 54,000 and 76,000 for Nogales and Santa Cruz County respectively, by 2035, or an annual growth rate of 5 percent per year (Udall Center

1993). Exact numbers are unknown for Nogales, Sonora, but it was estimated to have a population over 100,000 in 1990 (Udall Center 1993).

Binational Nature of the Problem

The border cities of Nogales, Arizona and Nogales, Sonora share not only a common border but also their water resources. Both cities depend upon the Santa Cruz Basin aquifer, which is bisected by the border, for their water supply. The aquifer is fed by the Santa Cruz River, which is also shared by the two countries. Some, but not all, threats to water quality in Santa Cruz County come from across the border, a characteristic that complicates attempts to alleviate water-quality problems.

Drinking Water Quality

The provision of safe, high-quality drinking water at affordable prices is crucial in maintaining the quality of life in all communities. To be viable, industrial, commercial, residential, and recreational uses of land require a dependable supply of drinking water. This dependency is particularly dramatic in rural areas since many communities are pursuing economic development strategies as a means toward higher incomes and full employment.

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

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

While implementation and enforcement of the SDWA has decreased health risks posed by drinking contaminated water, these actions also have dramatically increased the cost of treatment and delivery. Compliance costs have been particularly burdensome for small systems. The Environmental Protection Agency (EPA) has estimated that compliance across all systems will cost 1.4 billion dollars annually, with many systems having to install new equipment. The impact of these costs on small-system viability is evidenced by the fact that 70 percent of recent SDWA violations have been by small systems (GAO Report 1992), and by forecasts that additional requirements will exacerbate compliance problems for these systems still further.

In September 1993, EPA's Office of Ground Water and Drinking Water submitted to Congress its "Administration Recommendations for Safe Drinking Water Act Reauthorization." The document included 10 major recommendations for revisions to the SDWA. For rural communities, congressional decisions on reauthorization are expected to dramatically impact not only the current viability of small public water systems but also the future plausibility of establishing new systems as needed.

The following sections describe the SDWA and discuss the SDWA Reauthorization. The paper concludes by identifying issues and needs posed by the potential implementation of the Act, particularly as this applies to rural areas dependent upon small public water systems.

Overview of the Safe Drinking Water Act

The Safe Drinking Water Act (SDWA) was enacted by Congress in 1974, and was amended most recently in 1986. The purpose of the Act is to ensure that drinking water supplied to the public is safe; that is, it is free from contaminants that could adversely affect human health. The EPA is the federal agency responsible for promulgating regulations to carry out the provisions of the Act. In particular, the EPA is required to set standards and identify treatment techniques for contaminants, establish requirements for monitoring water

quality, and ensure the proper operation and maintenance of water systems (Fennemore 1992). Under the terms of the Act, water suppliers are responsible for making sure that the drinking water meets EPA standards, and for complying with established monitoring, operation, and maintenance protocols. However, it is important to note that the SDWA does not provide funding to support mandated treatment activities.

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

There are three major types of requirements in the SDWA: (1) sampling and reporting, (2) record-keeping, and (3) public notification (EPA 1993). Each supplier of water must collect samples from the water system, take them to a certified laboratory for analysis, and send the results to ADEQ. The laboratory results, name of the person(s) who collected the samples, dates and locations of sampling points, steps taken to correct problems, sanitary survey reports, and other information must be kept on file by the water supplier. Finally, any time there is a violation of a requirement, the public must be notified.

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

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

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

Sampling, Record-Keeping, and Public Notification

The legislative intent of the SDWA is to ensure that drinking water poses minimal risks to public health. The EPA is charged with implementing the intent of the Act and does so in two steps. First, "safe" drinking water is defined during the process of setting MCL standards and establishing treatment techniques. That is, drinking water that is in compliance with MCL standards and treatment requirements is judged to be safe for public health purposes. Second, protocols for sampling, record-keeping, and public notification have been established to promote the compliance of public water systems with EPA treatment and MCL safety standards. A brief description of these protocols follows.

In general, sampling requirements are detailed and complex, but the overall intent is to tailor sampling procedures to the type of containment being analyzed. The requirements address the types of analyses to be performed, the frequency of sampling, the location within the water system where sampling must occur, preservation techniques, transportation precautions, and laboratory certification. Some public water systems (e.g., small water systems, such as systems serving less than

3,300 people; or transient, non-community systems, such as systems serving hotels or restaurants) may receive variances or exemptions to these requirements.

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

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

ADEQ and the State's Primacy Role

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

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

A compliance status of 2 indicates that a system is in substantial compliance with the Act's provisions and is considered to pose only moderate risks to public health. Operational and maintenance procedures are most often the cause of violations; however, sampling procedures are also potential candidates. Examples of level-two violations include inadequate site cleanup, lack of proper fencing and security, susceptibility of the system to freezing, or user complaints. The issuance of an administrative order is the generally accepted practice for redressing substantial compliance violations.

A compliance designation of 3 denotes a system in noncompliance. Systems in noncompliance may have exceeded an MCL, failed to properly implement treatment requirements, ignored operation and maintenance procedures, or simply failed to sample the water. Examples of an operational and maintenance violation that would result in a rating of 3 include failing to install a pressure gauge, failing to chlorinate when necessary, or not having the well site graded properly. Initially, correction of a violation is addressed through an administrative order. If the system fails to comply, the order may proceed to full closure of the system.

SDWA Cost Implications for Small Public Water Systems

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

While additional capital costs may be avoided by small systems in Santa Cruz County, the same cannot be said of sampling costs. The 1986 amendments called for regulating an additional 66 contaminants by 1989 and for further expanding the

number of MCLs by 25 every year thereafter. The sampling costs associated with this regulatory expansion are likely to become progressively more burdensome if small systems are to comply.

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

Reauthorization of the Safe Drinking Water Act

In the 1993 version of the reauthorization, the EPA submitted to Congress an extensive list of proposed revisions to the SDWA (Trager et al. 1994). The original reauthorization bill, sponsored by Senator Max Baucus, included many of the recommendations proposed by the EPA, but failed to gain bipartisan support due to a failure to address the MCL standard-setting process. A second bill was subsequently introduced requiring increased risk/benefit analysis in standard setting, but was opposed by the EPA on the grounds that the mandated analysis would unrealistically delay the issuance of regulations. The 1995 Senate Bill (S. 1316), which was approved by a 99–0 margin but still awaits House approval, incorporated the more stringent risk/benefit analysis as well as addressing other public water system concerns (Environmental Law Update, October 1995). Ultimately, successful legislation will have to marry a variety of concerns. Particularly prominent among these concerns is the task of safeguarding public health while limiting the financial burden imposed by unfunded federal mandates.

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

Of direct concern to small public water systems are the following EPA recommendations (all addressed by S. 1316):

- Establishing (and adequately funding) a Drinking Water State Revolving Fund to provide low-interest loans to help water systems meet the costs of SDWA compliance.
- Requiring state-implemented programs to assess the viability of existing small systems and to prevent the formation of new, nonviable systems while restructuring and/or consolidating nonviable, noncompliant small systems.
- Establishing a less-expensive, "best available technology" that small systems could use to comply with the SDWA if they would not otherwise be able to achieve compliance through restructuring.
- Allowing for longer compliance deadlines for making drinking water standards effective, moving from 18 months after EPA promulgation to 60 months. S. 1316 also authorizes the EPA to grant variances to small public water systems serving fewer than 10,000 persons.

Noticeably absent from the list of EPA recommendations are suggestions for revising the standard-setting process itself. Between 1974 and 1986, the EPA issued rules regulating 23 drinking water contaminants. The 1986 SDWA amendments required the EPA to establish national drinking water standards or treatment techniques for 83 contaminants by 1989, and for 25 additional contaminants every three years thereafter. Critics of the program argue that this regulatory expansion places a disproportionate burden on small public water systems in rural areas by failing to account for local health risk impacts and budgetary realities. That is, failure to base drinking water standards on a site-specific, risk-benefit basis runs the risk of burdening drinking water programs to the point of collapse. In response to these criticisms, S. 1316 requires that the EPA balance benefits and costs before promulgating a regulation for a new contaminant.

SDWA Compliance in Santa Cruz County

The authors analyzed the compliance status of public water systems in the Santa Cruz Active Management Area (AMA) with respect to the

SDWA (see Figure 2). An AMA is an area, designated by the Arizona Groundwater Code, which needs water management programs to limit groundwater use. Data from the ADEQ were used to determine the compliance status of public water systems from the most recent survey completed in June 1994 (ADEQ 1994). Eighty-six percent of the population in Santa Cruz County uses public water systems. Private water systems, which comprise the remaining 14 percent, are not required to keep records or report to the ADEQ.

Of the 26 public water systems in Santa Cruz County, 81 percent are classified as very small (fewer than 1,000 persons served), 15 percent are classified as small (1,000–3,300 persons served), and 4 percent are classified as medium (3,301–50,000 persons served) (Figure 3). Only four of the 26 systems are in full compliance with the SDWA. Of these four, one system is very small and three are small.

System size does appear to affect the probability of being in compliance or noncompliance with SDWA regulations (Figure 4). There are no large public water systems in the Santa Cruz AMA. Only one medium-sized system operates in the county and it is in substantial compliance with the SDWA. There are four small systems, three of which are in full compliance and one which is in noncompliance. Of the 21 remaining very small systems, one (5%) are in full compliance, two (9%) are in substantial compliance, and 21 (86%) are in noncompliance.

In terms of bacterial contamination, seven systems were not in compliance with the SDWA. MCL violations occurred from a minimum of one to a maximum of three months in each of these seven systems. All of the bacterial violations occurred in very small systems, which accounted for 18 percent of all the systems in Santa Cruz County (Figure 5). Two of the seven systems in noncompliance were in acute violation, meaning that the coliforms found in the water samples contained fecal matter, which can be from animal or human sources. The health effects of fecal coliform microbiological contamination can include gastrointestinal illness, dysentery, hepatitis, typhoid fever, cholera, giardiasis, and cryptosporidiosis.

In one case, a very small system, inorganic MCLs were exceeded for nitrate contamination (Figure 6). Nitrates can cause methemoglobinemia, also known as blue baby syndrome.

Fifty-eight percent of the public water systems were in violation of SDWA regulations because of failure to comply with proper sampling procedures (Figure 6). Thirty-five percent of the water systems were out of compliance because they did not have an appropriately certified operator, while 19 percent had major operation and maintenance deficiencies. Failure to comply with SDWA standards does not necessarily mean increased health risks, but more likely an inability or unwillingness to meet the increasing costs associated with compliance.

Results from Santa Cruz County may not be unique to this area. Similar results were found from a study of eastern Pima County, where 50.5 percent of the small public water systems were in noncompliance with SDWA standards. Two percent of these small systems failed to meet MCL standards, and posed serious risks to human health. Sixty-four percent of the small systems failed to comply with operator certification, 57.1 percent failed to comply with sampling procedures, and 17.8 percent failed to comply with operation and maintenance standards (Cory and Moy 1994).

Statewide, of the 1,540 public water systems in Arizona that served 1,000 people or fewer in 1990, 826 or 54 percent were in noncompliance, with at least one major violation (Arroyo 1991, p. 3).

Potentially dangerous contaminants have been detected in Arizona public water systems, and over half of the smaller systems in both Pima and Santa Cruz counties are not consistently sampling their water with the regularity that is required to protect public health. This presents a threat to the public health. In those instances, the safety net established by the SDWA to protect the public is not working.

Funding for Small Systems

While additional financial resources would enable greater compliance, borrowing money is not even an option for most small water systems. Lending institutions are not inclined to loan money to systems that service so few customers. In addition, small systems have a limited credit base and limited market recognition, further reducing their chances of receiving loans. The same conditions that establish a small system as financially needy also undermine their efforts to receive funding. Small systems can usually only receive loans at high interest rates.

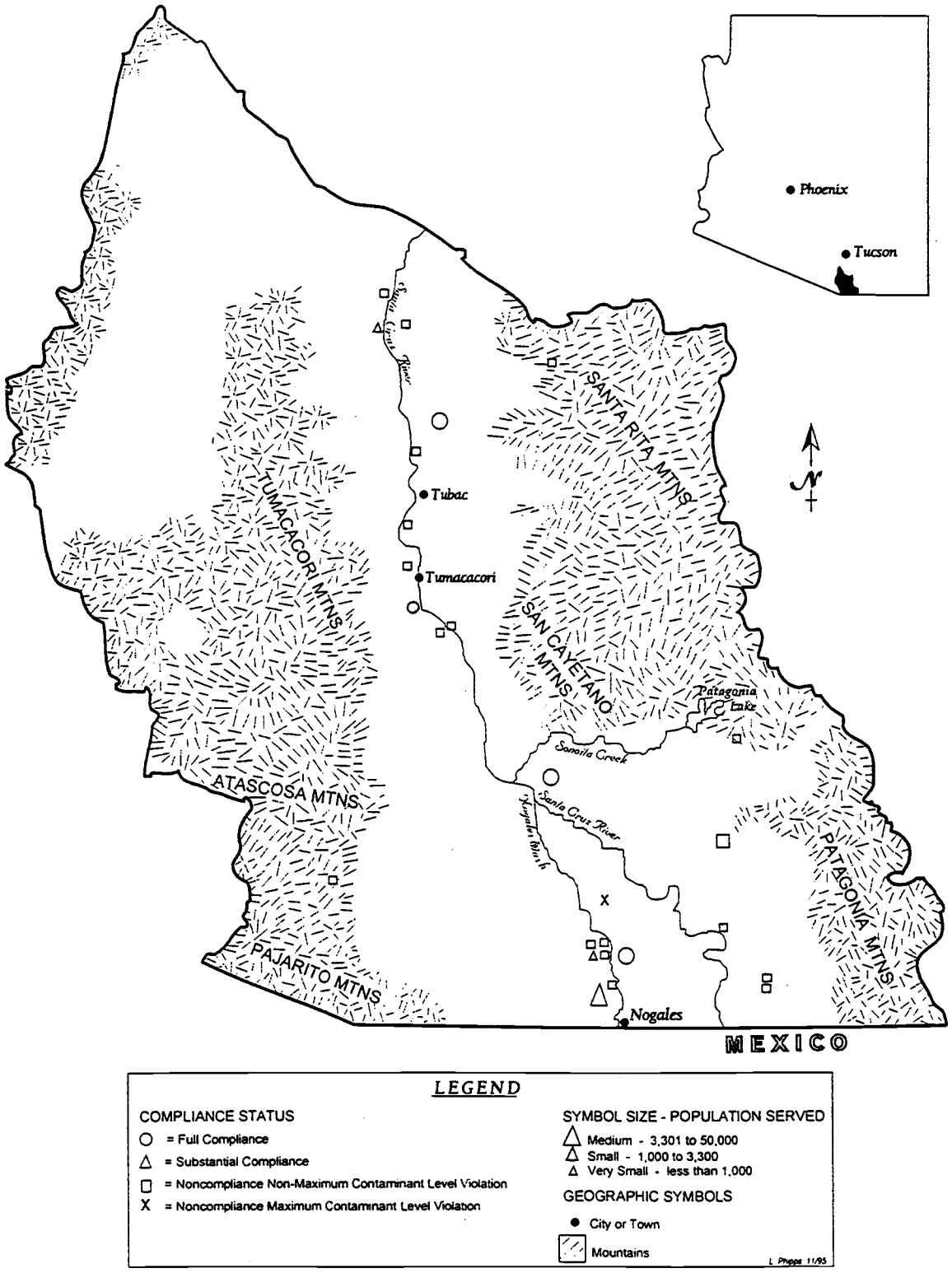

Figure 2. Santa Cruz active management area, public water systems.

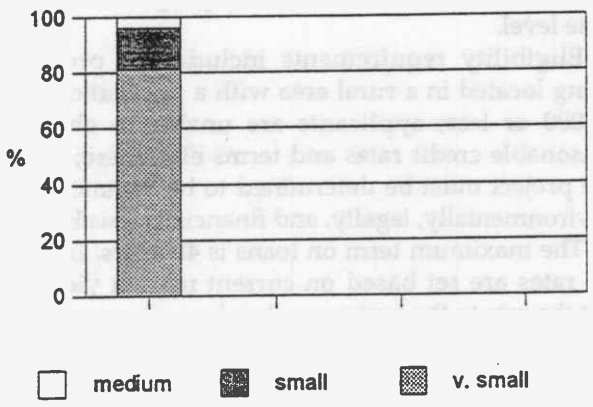

Figure 3. Water systems by size, Santa Cruz AMA.

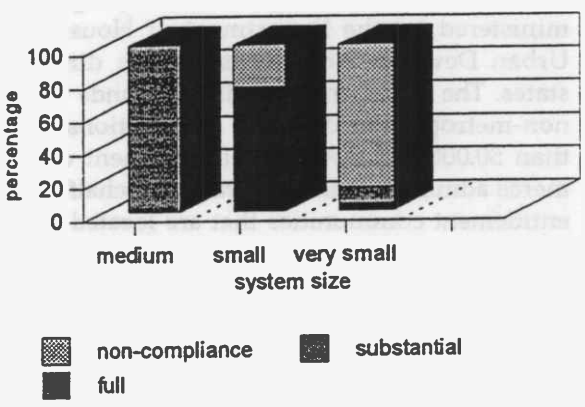

Figure 4. Compliance status and system size.

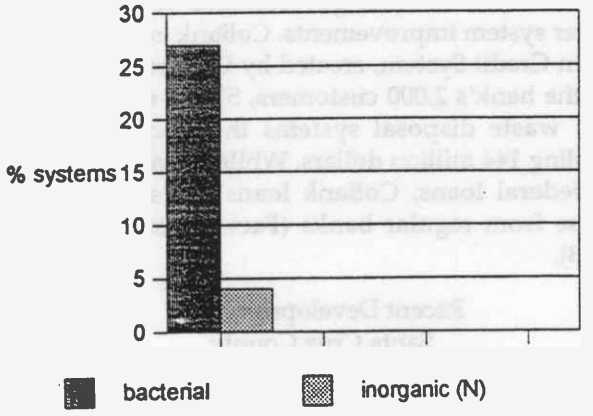

Figure 5. SDWA noncompliance, bacterial and inorganic.

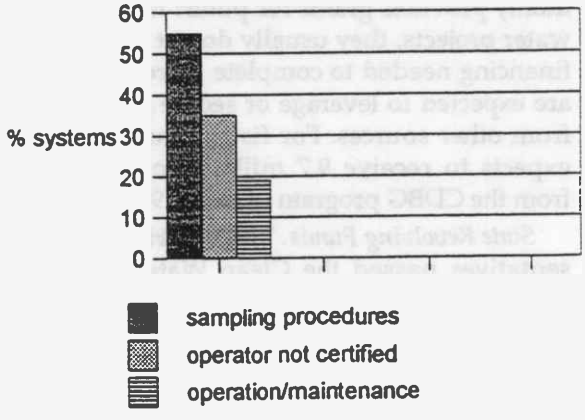

Figure 6. SDWA noncompliance, sampling and operation.

Funds are available to municipal water systems for small cities and towns through federal, state, and private institutions.

State Funding for Drinking Water Systems. The States and Small Cities program of the Community Development Block Grant (CDBG) program administered by the Department of Housing and Urban Development grants awards directly to states. The states, in turn, allocate funds to small non-metropolitan counties (populations of less than 50,000). The Arizona Department of Commerce administers the program on behalf of non-entitlement communities that are located outside of Maricopa and Pima counties, excluding the city of Yuma. Funds are allocated to the following planning regions or councils of governments: Northern Arizona Council of Governments for Coconino, Apache, Navajo, and Yavapai counties; Western Arizona Council of Governments for La Paz, Mohave, and Yuma counties; Central Arizona Association of Governments for Gila and Pinal counties; and South Eastern Association of Governments for Cochise, Graham, Greenlee, and Santa Cruz counties (ADEQ 1995).

The average grant amount for the States and Small Cities Program is $250,000 to localities with populations that average fewer than 5,000 persons. Improvements in public facilities are the most frequently funded projects, representing 54 percent of all funding. Water, sewer, and flood protection projects represent the largest share of public facilities funded. Although the state of Arizona commonly provides grants for public facilities such as water projects, they usually do not provide all the financing needed to complete a project. Localities are expected to leverage or secure grants or loans from other sources. For fiscal year 1995 Arizona expects to receive 9.7 million dollars in grants from the CDBG program (ADEQ 1995).

State Revolving Funds. The U.S. House of Representatives passed the Clean Water Act amendments in the summer of 1995 authorizing the use of state revolving fund (SRF) monies for communities to increase water use efficiency for projects with the goal of protecting water quality. Loans from the SRF have extended repayment periods for loans made to disadvantaged communities, and SRF monies have been used for technical, planning, and other related assistance to small systems (Arizona Environmental Law Update, July 1995). As of December 1995 the measure has not yet become law.

Federal Funding. The Rural Development Administration Water and Waste Disposal program is authorized to finance water systems in small municipalities with populations of 10,000 or fewer, and nonprofit organizations. The Farmers Home Administration administers the program at the state level.

Eligibility requirements include the project being located in a rural area with a population of 10,000 or less; applicants are unable to obtain reasonable credit rates and terms elsewhere; and the project must be determined to be technically, environmentally, legally, and financially feasible.

The maximum term on loans is 40 years. Interest rates are set based on current market yields, but the rate to the borrower also depends upon the median household income of the service area. Borrowers can receive one of three interest rates, the lowest being 4.5 percent for areas with low incomes and an existing water contamination problem.

Arizona Small Utilities Association. The Arizona Small Utilities Association, funded by the National Rural Water Association and the EPA, provides technical advice, but no funding, to rural water systems with populations of fewer than 10,000 people that lie outside of incorporated cities.

Private Lending Institutions. Many private lending institutions will fund drinking water system improvements, with the major drawback that interest rates are higher than federally backed loans. CoBank, or the Rural Cooperative Bank, makes loans to local governments for drinking water system improvements. CoBank is part of the Farm Credit System, created by Congress in 1917. Of the bank's 2,000 customers, 57 are rural water and waste disposal systems that receive loans totaling 144 million dollars. While not as attractive as federal loans, CoBank loans are superior to those from regular banks (Facts About CoBank 1993).

Recent Developments in Santa Cruz County

Plans for New WWTP in Santa Cruz County. The Environmental Protection Agency held a focus group meeting in September 1995 to incorporate public input into the decision making process for selecting a site for the new Nogales wastewater facilities project. The new treatment plant will help address transborder water quality problems and could in fact be located on the Mexico side. Com-

munity outreach activities are being coordinated by the EPA, the city of Nogales, the International Boundary and Water Commission, the ADEQ, and the South Eastern Arizona Association of Governments. The facilities planning process will take one year to identify alternatives and assess feasibility, at which point the environmental review process will begin.

Border Environment Cooperation Commission Begins Certifying Water Projects. The Border Environment Cooperation Commission (BECC) was created in accordance with the North American Free Trade Agreement (NAFTA) environmental side accords. Its mission is to address environmental problems in the U.S.–Mexico border region and it began to certify water projects in September of 1995 (Sprouse and Brown 1995). The BECC's projects deal with potable water, and a water supply and distribution project, with a projected cost of 42 million dollars, for Nogales, Sonora, was approved by the board in January 1996 (Tucson Citizen, January 19, 1996). The BECC and its sister institution, the North American Development Bank, provide another source of funding to help improve overall water quality both in Sonora, Mexico and in Santa Cruz County.

Future Outlook

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

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

To address financing problems faced by small water systems, the EPA in 1990 suggested a multi-community cooperative arrangement approach in a report titled Environmental Pollution Control Alternatives: Drinking Water Treatment in Small Communities. Under this type of arrangement, small communities would share resources with other communities, large or small, through a regional water supply authority. The benefits of such an arrangement would make use of economies of scale and result in increased cost effectiveness, better water quality, and more efficient operation and management. Some of the types of cooperative approaches available to small water systems include the following.

- Centralization of functions. By working together, a group of small systems would be able to centralize purchasing, maintenance, engineering services, laboratory services, and billing. In addition, several small systems could better afford resources, such as full-time, highly skilled personnel, by having them work part time at different locations.
- Interconnection of existing systems. To take advantage of economies of scale, two small systems can be interconnected, or a large system with a small system. The only restriction is that the two systems must be in close physical proximity.
- Utilization of satellite facilities. Smaller systems sometimes can access the resources of larger systems, even without a physical or economic connection to the larger facility. These satellite facilities have access to resources such as technical, managerial, or operational assistance; operation and maintenance responsibility; or wholesale priced treated water.
- Establishment of water districts. Water districts, formed by county officials, can combine existing districts and physically connect water systems. By creating a water district, privately owned systems can collectively become eligible for public grants and loans, when available.
- Creation of county or state utilities. Within its boundaries, a county or state government can appoint a board to construct, operate, and maintain a water system. The advantage of such an approach is that bonds and property taxes may be used either to construct a new facility, or to upgrade an existing one.

While a great deal of uncertainty remains about the net impact of the EPA's recommendations to revise the SDWA, and the reauthorization debates in Congress, one effect on consumers of drinking water in rural areas is inescapable: as consumers demand higher quality water and as stricter drinking water standards are implemented, public water systems will be forced to charge higher

prices. New and creative approaches, such as those mentioned above, will have to be considered by small water systems to help keep costs down while complying with public health regulations.

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