

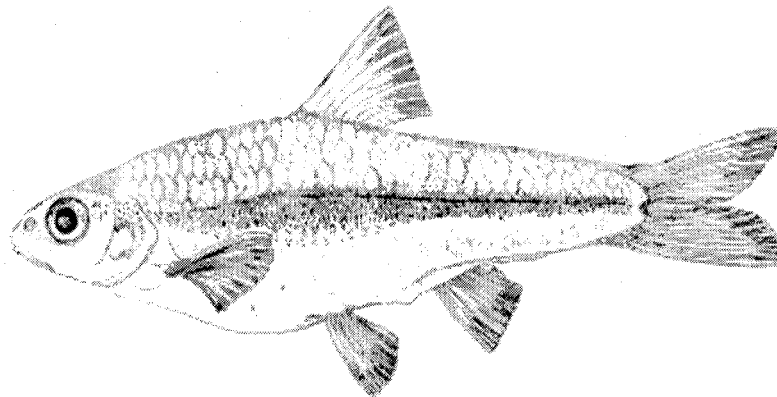
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AQUATIC FAUNA OF THE NORTHERN CHIHUAHUAN DESERT



CONTRIBUTED PAPERS FROM A SPECIAL SESSION WITHIN
THE THIRTY-THIRD ANNUAL SYMPOSIUM OF
THE DESERT FISHES COUNCIL

HELD

17 NOVEMBER 2001

AT

SUL ROSS STATE UNIVERSITY, ALPINE, TEXAS

EDITED BY

GARY P. GARRETT AND NATHAN L. ALLAN

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Aquatic Fauna of the Northern Chihuahuan Desert

Contributed Papers from a Special Session within the Thirty-Third
Annual Symposium of the Desert Fishes Council

GARY P. GARRETT AND NATHAN L. ALLAN

Heart of the Hills Fisheries Science Center, Texas Parks and Wildlife Department
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AQUATIC FAUNA OF THE NORTHERN CHIHUAHUAN DESERT

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FORWARD

The twelve papers presented in this volume are a collection of proceedings from a special session within the 33rd Annual Symposium of the Desert Fishes Council (DFC), held on November 17-20, 2001, at Sul Ross State University, Alpine, Texas. It was the desire of the local DFC planning committee (co-chaired by Nathan Allan and Chris Hoagstrom) that the meeting should have a session dedicated to the issues of special conservation concern for the region. We invited speakers from a variety of disciplines with expertise on aquatic issues in the northern Chihuahuan Desert. Our intent was to expand the scope of interest from fishes to include other aquatic biota and to provide context for conservation considerations.

We felt strongly that the information presented at the special session must be captured in a proceedings document to provide a single source of written information on aquatic biota of this region. No volume could be considered a comprehensive assessment of such a vast topic, but we intended to include as much information as possible to provide a realistic overview of the biota of the region and the conservation issues we face. The papers are clearly biased towards fishes, because of our bias as fishery biologists. We are keenly aware that plants, amphibians, reptiles, snails and other groups are equally dependent on the waters of the desert and their conservation is equally significant. All of the invited speakers were offered the opportunity to contribute their manuscripts to this publication and 12 of the 14 chose to do so.

The geographic extent of the "northern Chihuahuan Desert" was not intended to be a well-defined boundary, but rather a general guide for the papers presented. Many of the papers are focused on the Trans Pecos area of West Texas. This is because of the experience of the authors solicited for inclusion, not necessarily any statement of priority of this area within the entire Chihuahuan Desert.

We are especially honored by the contributions of two of the eminent biologists of the northern Chihuahuan Desert, Dr. Clark Hubbs and Dr. James F. Scudday. Their early work and insight helped build the groundwork for most of us today.

Dr. Hubbs began his work at the University of Texas at Austin in 1948 and is now Professor Emeritus. He remains an active researcher and continues to contribute to the conservation of fishes. In recent years his emphasis has been primarily on spring fishes and correlates with chemical parameters. As with so much of his previous endeavors, this work will very likely provide a baseline for many researchers for years to come.

Dr. Scudday retired as a biology professor from Sul Ross State University in 1996, after 33 years on the faculty. A native of Fort Stockton, Texas, Dr. Scudday is a notable authority on vertebrates of Trans Pecos, Texas, largely because of his lifetime of experience there. Although much of his paper is anecdotal, the picture he paints of the Trans Pecos area more than 50 years ago is a stark contrast to the area today and provides an important perspective that is relevant to the remainder of the volume.

This volume also includes a number of esteemed scientists who have spent decades researching zoology, ecology, and geology in the Trans Pecos and adjacent regions. A primary goal was to assemble a large amount of the institutional knowledge currently available for the region. If nothing else, this volume should serve as a valuable source for information on a wide array of topics related to aquatic resource conservation and should stand as a useful literary reference for future investigators.

We wish to offer our sincere appreciation to everyone who volunteered his or her time and expertise to help complete this document. A special thanks to Dr. Gary Garrett for serving as the managing editor of the proceedings--we doubt he had any idea what he was getting into when he took the job. All the papers have been peer reviewed by at least three other persons. All of the authors also served as reviewers for other papers. We also thank Tim Bonner (Southwest Texas State University), Ray Mathews (Texas Water Development Board), Gordon Linam (Texas Parks and Wildlife Department), and Kirk Winemiller (Texas A&M University) for serving as reviewers of various papers.

Thanks also to the DFC Executive Committee (especially Dean Hendrickson and Phil Pister) for giving us latitude to work "outside the box" of the normal DFC meetings. We appreciate the Museum of Texas Tech University (especially Robert Baker and Jackie Chavez) for being willing to include this document within the Special Publication series. Finally, we recognize the organizations (and the people who made it happen) which provided financial support for both the meeting in Alpine and this publication: West Texas Program Office of The Nature Conservancy (John Karges); World Wildlife Fund, Chihuahuan Desert Program (Jennifer Atchley); Desert Fishes Council (Phil Pister); Chihuahuan Desert Research Institute (Cathryn Hoyt); and U.S. Fish and Wildlife Service, New Mexico Fishery Resources Office (Jim Brooks) and Austin Ecological Services Office (Bill Seawell).

Nathan Allan
Christopher Hoagstrom
Co-Chairs, Local Committee, DFC 2001

GROUNDWATER SYSTEMS FEEDING THE SPRINGS OF WEST TEXAS

JOHN M. SHARP, JR., RADU BOGHICI, AND MATTHEW M. ULIANA

ABSTRACT

Major existing and former springs of the northern Trans-Pecos, Texas, include the Balmorhea Springs (San Solomon, Phantom Lake, Giffin, and East and West Sandia) in Reeves and Jeff Davis Counties and Comanche Springs, Leon Spring, and Diamond-Y Springs in Pecos County. Understanding the regional groundwater flow systems that feed or fed these springs is needed to manage regional water resources, including the springs that provide islands of aquatic habitat. Some springs have ceased to flow or now flow at greatly diminished rates. Data indicate that spring discharges have been gradually declining for at least the last 100 years. In addition, groundwater extraction for municipal, domestic, and irrigation uses threatens continued spring flows. The individual groundwater basins are connected through regional

flow systems in fractured, karstic carbonate rocks. Regional fracture trends connect the major recharge and discharge areas and localize discharge from carbonate aquifers. Analysis of fracture systems allows interpretation of regional flow systems and regional-scale permeability. Recharge is from fractures in the highlands, losing streams on proximal portions of alluvial fans, irrigation return flow, and interbasin flow. Discharge is to the springs, by wells, and in the past to the Pecos River. $^{87}\text{Sr}/^{86}\text{Sr}$ ratios and other chemical and isotopic data confirm the inferred regional flow systems and suggest that some of the springflow recharged during the Pleistocene. The groundwater system is evolving because of both climatic trends and anthropogenic effects.

INTRODUCTION

Trans-Pecos Texas encompasses the general area of Texas west of the Pecos River (Figure 1) and is the most southeastern portion of the Basin and Range physiographic province in the United States. It has a subtropical semiarid climate. Average annual precipitation is less than 300 mm, and precipitation increases with increasing elevation (e.g., Schuster, 1996). With the exception of a few significant springs and the brackish Pecos River and Rio Grande, surface water resources are minimal. A number of individual groundwater basins form parts of regional groundwater flow systems. Regional-scale structural features create a template for fractures and karst features that control the flow systems. The regional flow systems, in turn, discharge at springs that provide unique wetland habitats for endangered aquatic species. Through wells and spring flow, these systems have been developed to meet much of the area's municipal, domestic, and agricultural needs.

Brune's (1981) compendium listed the important springs of the region. This study concentrates on those in the northern Trans-Pecos associated with endangered species and the groundwater systems associated with these springs. Figures 1 and 3 show the locations of the springs. The unique biotas that inhabit the spring systems suggest persistent, long-term springflow discharge. However, data indicate that discharge from these springs may have been in decline for the past 100 years and, possibly, for the past several thousand years (Hall, 1990; Sharp et al., 1999; Musgrove, 2000) as is this case for most of the southwestern U. S. A.

Increased groundwater pumpage for municipal and agricultural use has hastened or caused springflow declines and changes in the regional flow systems. Proper management of the groundwater in this region is important to maintain spring flows and the habitats

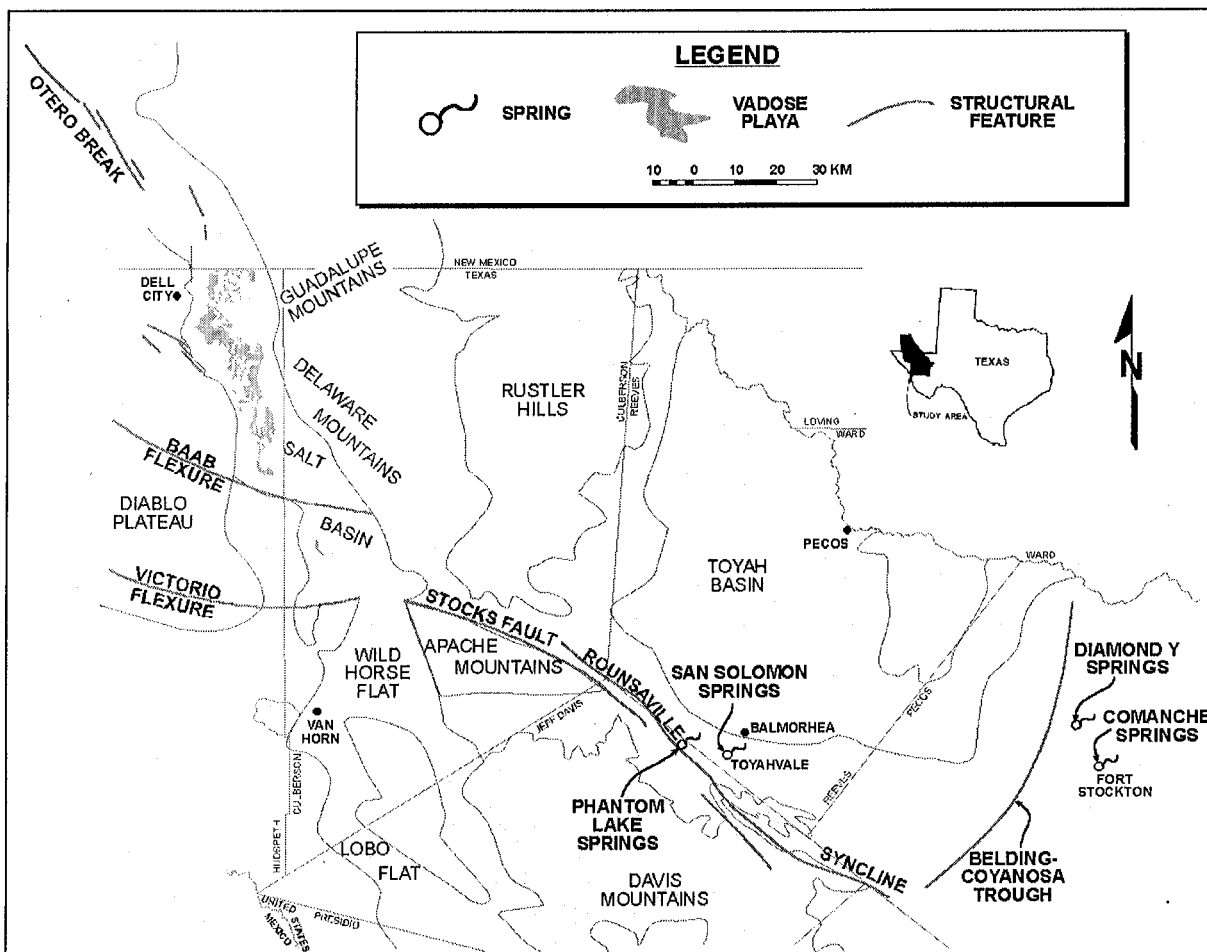


Figure 1. Study area is located in northern Trans-Pecos Texas and southeastern New Mexico. The Salt Basin, Wildhorse Flat, Lobo Flat, and the Toyah Basin are clastic basin fills; the Davis Mountains are volcanic rocks. The major springs are delineated. Giffin Spring is adjacent to San Solomon Springs in Toyahvale and East and West Sandia Springs are in the town of Balmorhea. Phantom Lake Springs is closely linked to the springs in Toyahvale. Leon Springs formerly discharged between Comanche and Diamond Y Springs.

for the endangered species and to meet the region's municipal and agricultural needs. Herein we review the regional hydrostratigraphy and structural geology and discuss the hydrostratigraphy, hydrogeology, structural

geology, regional hydrogeology, fracture controls on permeability, some springwater chemistry, and implications for spring discharge and water resource management.

HYDROSTRATIGRAPHY

Geologic units range in age from Precambrian to Holocene, but the most significant units hydrogeologically are: 1) the Permian shelf, reef, and basinal sediments of the Delaware Basin; 2) Cretaceous carbonates; 3) Tertiary igneous rocks in the Davis and Barilla Mountains; and 4) Cenozoic alluvium. Permian rocks are subdivided into three hydrostratigraphic facies (Figure 2) with highly different hydraulic properties (Hiss, 1980; Nielson

and Sharp, 1985; Boghici, 1997; Mayer and Sharp, 1998; Boghici and van Broekhoven, 2001). The Guadalupian shelf margin (reefal facies) provides excellent aquifers as exemplified by the Capitan aquifer, which has high porosity and permeabilities that are the result of extensive karstification (e.g., Carlsbad Caverns and associated caves in southeast New Mexico, the Apache Mountains, and the Glass Mountains). Aquifers in the Per-

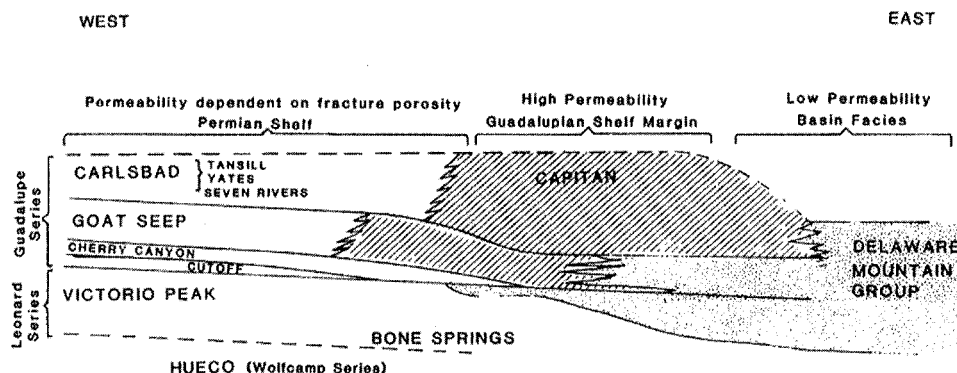


Figure 2. Permian hydrostratigraphic facies (after Nielson and Sharp, 1985). The stippled pattern denotes sandstones and carbonates overlying basinal evaporite-rich rocks; the crosshatched pattern denotes highly permeable reefal facies rocks of the shelf margin.

mian shelf facies have highly variable fracture-dependent permeability. Karstification is controlled by the fracture set characteristics (e.g., the Bone Spring aquifer that supplies Dell City). The basin fill facies rocks, including the Rustler Formation generally possess low permeability and extensive evaporite deposits. They form aquifers with low permeabilities, poor quality water, and low well yields. The Rustler provides brackish water to wells and along a deep fault system to Diamond Y Springs.

Lower Cretaceous rocks crop out in the eastern, southeastern and southwestern parts of the Toyah Basin and on the Diablo Plateau (Figure 1). These rocks represent mostly marginal, near-shore, and marine facies. They supply irrigation and livestock wells with fresh to slightly brackish water (Ogilbee et al., 1962; Boghici, 1997). The Cretaceous Edwards-Trinity is the most important aquifer in the Diamond Y area. It underlies most of Pecos County, as well as parts of Reeves, Culberson, and Jeff Davis Counties (Anaya, 2001). The more permeable units in the Edwards-Trinity are the lower Cretaceous sands and limestones, which are hydraulically connected with the overlying Pecos Cenozoic alluvial aquifers of the Coyanosa and Toyah Basins. In some locales, Cretaceous carbonate units are juxtaposed with Permian reefal rocks and form parts of the same flow system, such as the one that flows to Balmorhea and the Toyah Basin (LaFave, 1987; Uliana and Sharp, 2001). Phantom Lake Springs issue from a cave opening in Lower Cretaceous limestone. The Cretaceous carbonates of the Diablo Plateau support a regional aquifer and a less extensive perched aquifer. These rocks can be extremely transmissive because of fracture and solution porosity (Scalapino, 1950; Kreitler and Sharp, 1990).

Groundwater flows to the northeast and discharges into the Salt Basin, where it evaporates in gypsum flats.

Tertiary igneous rocks are mostly ash-flow tuffs and lava flows that overlie the Cretaceous rocks. The volcanic rocks of the Davis Mountains do not contain significant regional aquifer systems (Hart, 1992; Chastain-Howley, 2001), but runoff from them contributes to recharge of the Toyah Basin alluvial aquifer (LaFave and Sharp, 1987; Uliana, 2000) and, presumably, other surrounding basins.

Thick Cenozoic fluvial siliciclastic deposits occur in the Toyah and Salt Basins. Other recent deposits include fluvial terraces, playa muds and evaporites, aeolian deposits, and colluvium. Aquifers in these units provide significant amounts of water to wells and municipalities in the Toyah Basin and in Wild Horse Flat (Gates et al., 1980; Sharp, 1989; Ashworth, 1990). The Toyah Basin was formed by dissolution of underlying Permian evaporites and is filled with up to 470 m of Cenozoic alluvium (Maley and Huffington, 1953). The alluvial sediments filling in the Salt Basin reach a thickness of over 750 m (Gates et al., 1980). Groundwater divides separate the basin into three different flow systems. The northern and the middle areas are closed basins with recharge occurring through the bounding faults on the east and west of the basin, and discharge through the gypsum flats or vadose playas in the center of the basin (Boyd, 1982; Nielson and Sharp, 1985). The Wild Horse Flat part of the basin, however, lacks playas. In addition, the oldest mapped potentiometric surface (Nielson and Sharp, 1985; Sharp, 1989) indicates interbasin flow through the Apache Mountains towards the east.

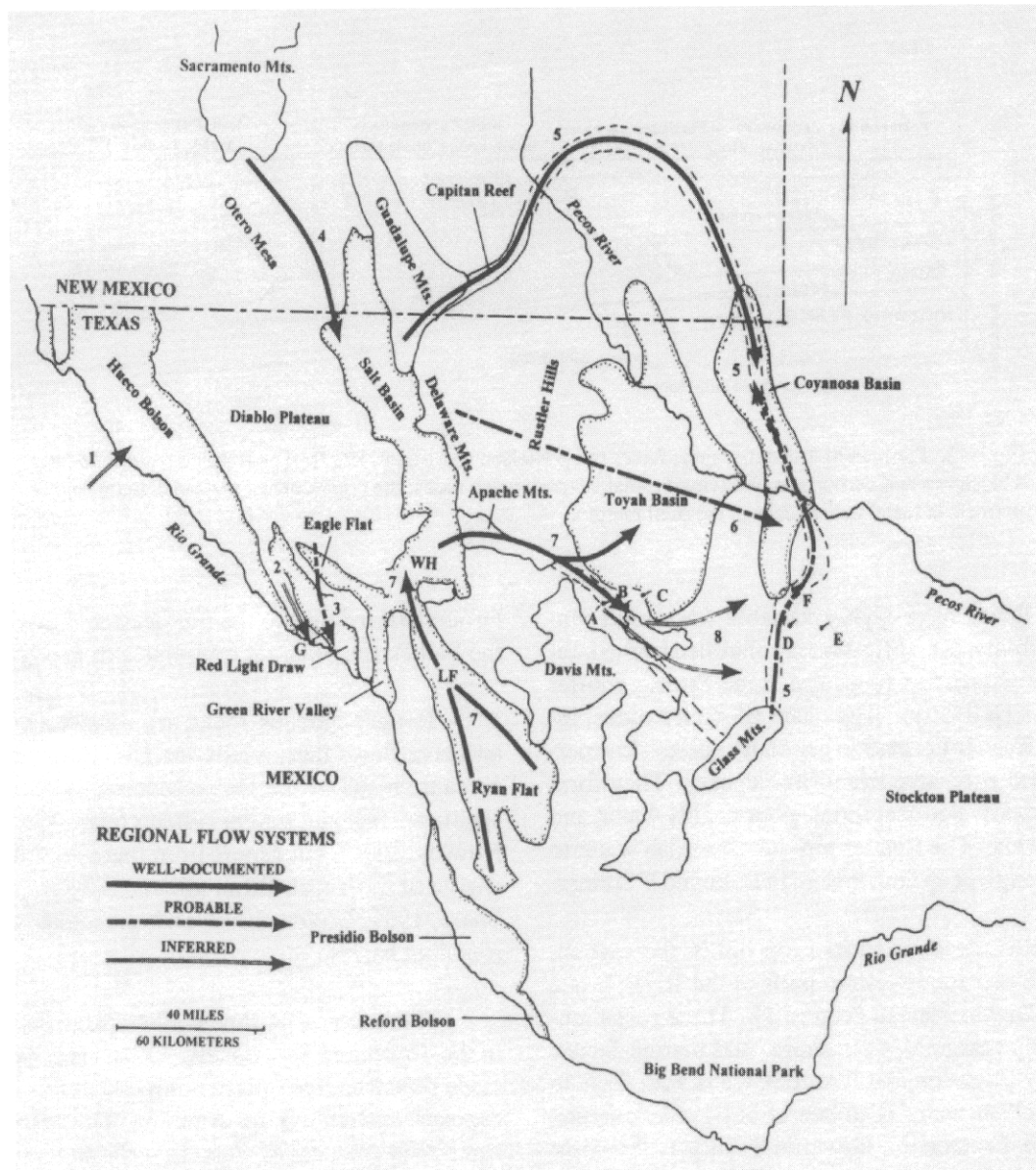


Figure 3. Regional flow systems of Trans-Pecos Texas (Sharp, 2001). WH and LF denote Wild Horse Flat and Lobo Flat of the Salt Basin. Springs are denoted by letters - A, Phantom Lake Spring; B, San Solomon and Giffin Springs; C, East and West Sandia Springs; D, Leon Springs; E, Comanche Springs; F, Diamon-Y Springs; and G, Indian Hot Springs. A, D, and E no longer flow. The regional flow systems are: 1 and 2, the discharge at the Fabens artesian zone and Indian Hot Springs (G), respectively; 3, Eagle Flat - Red Light Draw flow system; 4, Sacramento Mountains - Dell City flow system; 5, flow systems in the Capitan Reef; 6, eastward flow in the Delaware Basin, perhaps discharging at Diamond-Y Springs (F); 7, the Salt Basin - Toyah Basin - Pecos River system that also feeds Balmorhea Springs (A, B, and C); and 8, speculative eastwards extensions of this last flow system.

STRUCTURAL GEOLOGY

Trans-Pecos Texas has been subjected to at least five major tectonic episodes that formed fault zones and structural trends that have been repeatedly reactivated throughout the history of the area. Precambrian compressional events generated the basic northwesterly

trends that are still predominant and that influenced subsequent structures. The early Pennsylvanian Ouachita collision was responsible for the formation of the Delaware Basin and the thick sequences of Permian sedimentary rocks that form a significant part of the re-

gional groundwater systems. The morphology of the Delaware Basin has influenced later fault patterns. For example, major faults in the Apache Mountains and the Guadalupe Mountains run parallel to the Permian paleoreef front. Later events include the Mesozoic rifting of the Gulf of Mexico, the early Cenozoic Cordilleran/Laramide

Orogeny, and Eocene Basin and Range extension. These events caused the repeated reactivation of the earlier structural features. Faulting is active today (Goetz, 1977). Details of the region's tectonic/structural setting are given in Dickerson and Muehlberger (1985) and Muehlberger and Dickerson (1989).

REGIONAL HYDROGEOLOGY

Groundwater production in Trans-Pecos Texas is concentrated in the Permian and Cretaceous rocks and in the Cenozoic alluvial fill in the Salt and Toyah basins (Davis and Leggat, 1965; Couch, 1978; Hiss, 1980; Ashworth, 1990; Sharp, 2001). Evidence exists for significant groundwater flow in fractures and in karst conduits (LaFave and Sharp, 1987; Mayer and Sharp, 1998; Uliana, 2000; Sharp, 2001). Both local and regional flow systems exist in the area. Figure 3 shows the major regional flow systems inferred for northern Trans-Pecos Texas. The major springs discharge both from regional and local flow systems associated with Permian/Creta-

ceous carbonate rocks (White et al., 1941; LaFave and Sharp, 1987; Uliana and Sharp, 2001). The flow systems that feed the currently active springs harboring endangered aquatic species (San Solomon, Giffin, East and West Sandia, and Diamond Y Springs) are only two on the documented, probable, or inferred regional flow systems in northern Trans-Pecos Texas and southeastern New Mexico. The other flow systems may have once discharged to springs (i.e., Crow Springs near Dell City) that have also have once harbored unique biota. All these regional flow systems are in fractured carbonate rocks.

FRACTURE CONTROLS ON GROUNDWATER FLOW

The regional flow systems are controlled by fracture systems. The density and orientation of fractures in the Trans-Pecos region reveal a definite relationship between the regional structural trends and the fracture orientations. LaFave and Sharp (1987) and Uliana (2000) document fracture orientations in the Apache and Delaware Mountains that follow the prevalent N10W orientation of regional structural trends. Mayer (1995) and Uliana (2000) used aerial photos and field studies to map lineaments and fractures. They document the correlation between fault orientations and the regional structural grain. Fault patterns in the Salt Basin (Goetz, 1977) show a similar correlation between fracture patterns and the regional structural grain.

Fracturing of the carbonate rocks influenced their subsequent karstification. Nielson and Sharp (1985), LaFave and Sharp (1987), and Uliana (2000) document regional connections between Wildhorse Flat and the Toyah Basin, including the springs at Balmorhea. Fracturing of Permian reefal rocks and karstification create a high permeability zone along the Stocks Fault-Rounsaville Syncline trend (Figures 1 and 3). Hydraulic head data show that water discharged from Wildhorse Flat at one

end of the structural trend and flowed into the Toyah Basin along its southwestern edge, near the springs. Additional evidence supporting the regional flow hypothesis includes the lack of discharging playas in the Wildhorse Flat section (Figure 1), fracture trends in the Apache Mountains, and isotopic data (Uliana, 2000; Uliana and Sharp, 2001). The northern parts of the Salt Basin contain extensive vadose playas that are primary natural discharge features. The lack of playas in Wildhorse Flat is consistent with groundwater discharge by interbasin flow through the Apache Mountains. Uliana and Sharp (2001) examined the $^{87}\text{Sr}/^{86}\text{Sr}$ distributions in Trans-Pecos groundwater. High $^{87}\text{Sr}/^{86}\text{Sr}$ values occur at the up-gradient end of the flow system; these are caused by groundwater interaction with Precambrian rocks (including clasts in alluvial fans) along the west edge of Wildhorse Flat. $^{87}\text{Sr}/^{86}\text{Sr}$ values decrease along the hypothesized regional flow paths. This suggests that the high $^{87}\text{Sr}/^{86}\text{Sr}$ values in the Wildhorse Flat groundwater equilibrate with the Permian and Cretaceous carbonates and fluid mixing with other waters recharged or displaced along the flow path. Phantom Lake Springs formerly issued from an opening in Cretaceous limestone. This opening leads into a network of caverns. Cave divers

have mapped and surveyed over 2300m and measured groundwater discharge in the caverns. Their map (Tucker, 2000, reproduced in Uliana, 2000, p. 21) indicates that the cave network follows a linear trend that parallels the Stocks Fault - Rounsaville Syncline trend. In January 1999, Phantom Lake Springs ceased to flow for the first time in probably at least 200 years (Figure 4).

A similar high-permeability structural trend (the Otero Break) connects the Sacramento River recharge area in southern New Mexico and the Dell City, Texas, irrigation district (Mayer and Sharp, 1998). This inferred conduit flow (Flow system 4 on Figure 3) is confirmed by both geochemical and hydraulic head data. This flow system formerly discharged at Crow Springs in the Salt Basin (Ashworth, 2001). Flow system 3 from Eagle Flat is presumed to discharge to the Rio Grande (Darling and Hibbs, 2001). Indian Hot Springs (G on Figure 3) discharge from carbonate rocks and the flow is through fractured carbonate rocks in the U. S. A., and, perhaps, Mexico. Although undocumented, similar regional flow systems may exist in southern Trans-Pecos Texas and Northern Mexico.

Comanche Springs (Figures 1 and 3) formerly issued from a 673-m long, fracture-controlled cave formed along a N60-65W trending joints (Boghici, 1997; Veni, 1991). Dye tracer tests conducted in the 1950's demonstrated that groundwater was flowing to Comanche Springs at rates of up to 3.2 km/day (Sparks, cited in Veni, 1991). A structural low in the permeable Cretaceous limestones (the Belding-Coyanosa trough) extends some 64 km to the southwest of Comanche Springs connecting them with the recharge areas in the vicinity of the Glass Mountains (Boghici, 1997; Boghici and Van Broekhoven, 2001).

Darling (1997), Darling et al. (1995), and Darling and Hibbs (2001) examined oxygen, deuterium, and carbon isotopes in the groundwater in the western portions of the study area (in particular, in flow systems 2 and 3 on Figure 3). These data suggest recharge during a cooler and wetter Pleistocene climate. Groundwater ages of 8,000 to 50,000 years are indicated. Balmorhea springs show little seasonal variation. Their waters are brackish and slightly higher than mean annual surface temperature. Isotopic data, although sparse, support Darling's results, indicating long residence times (Uliana, 2000;

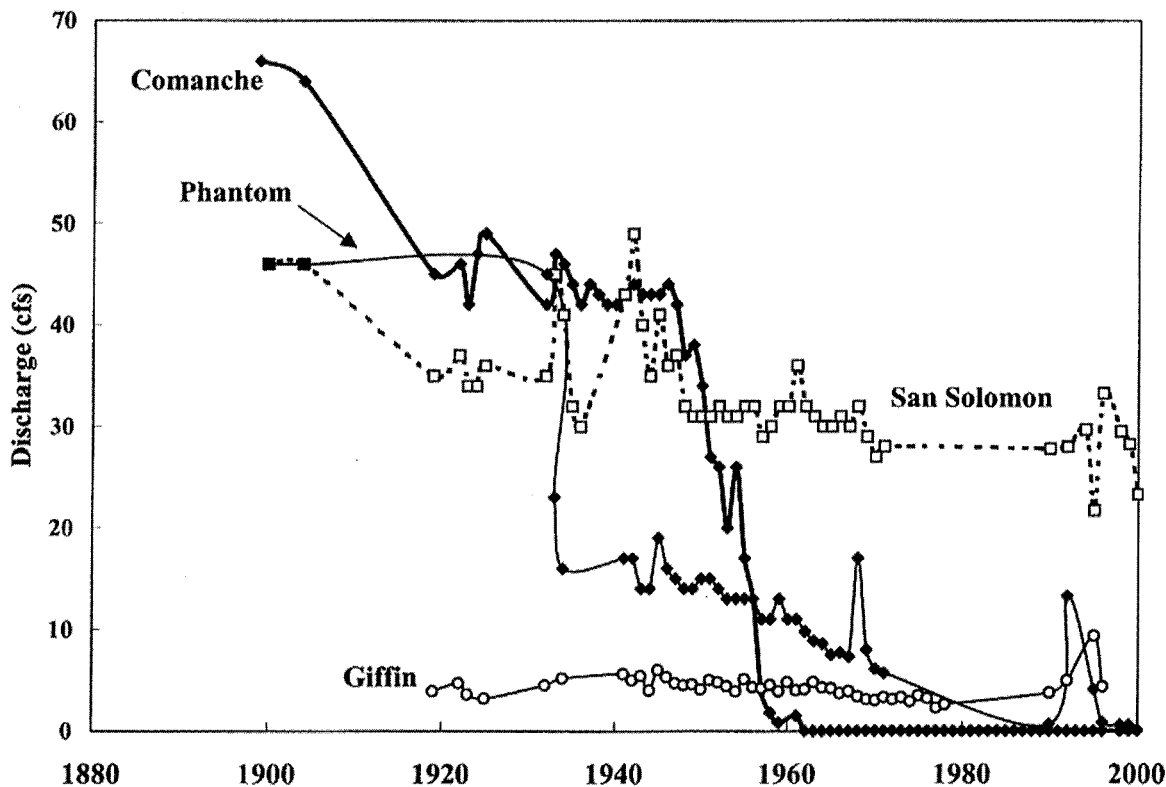


Figure 4. Spring flow hydrographs of Comanche, Phantom Lake, San Solomon, and Giffin Springs. Data are from U. S. Geological Survey, U. S. Bureau of Reclamation, and Groundwater Field Methods Class (1990, 1992, 1995, and 1996).

Uliana and Sharp, 2001). In addition, the unique spring ecosystems suggest that spring flow and spring chemistry have remained relatively stable for extended periods of time. Such stability in a semi-arid zone implies a re-

gional flow system because local flow systems show greater seasonal variations. Consequently, a few years of drought would not be suspected to impact spring flows significantly.

SPRING DISCHARGE AND WATER RESOURCE MANAGEMENT

Figure 4 shows the discharge of the four of the largest springs in the area. Early (before 1920) data are less reliable, but several trends are evident. First, springflows appear to have been declining prior to the development of extensive irrigation in the 1940's. This is consistent with longer-term climatic studies that suggest a drying of the region (cited in Kreitler and Sharp, 1990). Second, there is variability in discharge, particularly, when average annual discharges are low. This is caused by normal climatic variability and may not be indicative of long-term sustained trends. The variations probably reflect short-term variability in groundwater recharge. Some later data on San Solomon, Giffin, and Phantom Lake Springs are point (in time) measurements taken by the Groundwater Field Methods Classes (1990, 1992, 1995, and 1996). These are not annual averages, but reflect the short term variability. Third, San Solomon and Giffin Springs show a remarkably steady annual flow regardless of climatic variability. Flow has been relatively constant since the 1930's even through the major drought of 1947-1955. Four, Comanche and Phantom Lake spring discharges began to decline at greater rates when groundwater pumpage for irrigation commenced. Comanche Springs ceased to flow about 1961, and Phantom Lake Springs largely ceased to flow in January 1999.

Balmorhea Springs.—San Solomon, Giffin, and Phantom Lake springs are in close proximity and are probably fed from the same fractured, karstic conduit (Uliana and Sharp, 2001). The conduit system, which is accessed at the Phantom Lake Spring orifice, was gauged in 1997 by Tucker (2000). His preliminary data showed that, at that time, only a small percentage of the flow in the conduit discharged at Phantom Lake Springs. The bulk presumably discharges to San Solomon and Giffin springs, but this is not yet documented. Phantom Lake Spring fed a *ciénega* and canal system that harbored endangered fish species. Inconsistencies in spring discharges fed from the fractured, karstic conduit trend are typical of karstic systems.

The decline in Phantom Lake springflow may be caused by several factors. First, a regional lowering of the water table is documented by comparing recent data from The University of Texas Groundwater Field Methods classes (1990, 1992, 1995, and 1996) and the U. S. Bureau of Reclamation with the earliest known data (White et al., 1941). These show that the water table near Balmorhea has been lowered by some combination of long-term climatic change and, probably more importantly, regional pumping for irrigation and municipal uses. This pumping could include both local well effects and the regional lowering of heads in the Toyah Basin. A second possibility is that the "plumbing" itself has changed because of either opening of fractures/conduits because of dissolution, tectonic activity, or changing sediment storage in the conduits that could increase or decrease the permeability of various branches of the conduit. Major pulses in turbidity of spring discharge have been observed (White et al., 1941; Kreitler and Sharp, 1990). Loss of recharge because of domestic wells in the Davis Mountains is another potential, but probably secondary, effect. Finally, the beheading of the regional flow system because of pumping in Wildhorse Flat may have an effect. Water recharging Wildhorse Flat now supplies the communities of Van Horn and Sierra Blanca, as well as local irrigators of cotton, pecans, hay, and vegetables (Nielson and Sharp, 1985; Kreitler and Sharp, 1990) so that discharge from Wildhorse Flat via interbasin flow has been diminished, if not stopped. At present, there are insufficient data to differentiate between these hypotheses or eliminate any of them.

Pecos County Springs.—The cessation of flow at Comanche and Leon Springs is closely correlated in time with the onset of irrigation pumpage, although, quality long-term discharge data are not available for Leon Springs. Diamond Y Springs, however, continue to flow. The discharge data for the Diamond Y Springs are sparse: only four measurements between 1943 and 1987, and only the main spring was gauged. Based on their re-

sponse to rainfall, Veni (1992) indicated the existence of two distinct groups of springs and seeps. The springs' response suggests slight to moderately extensive flow conduits feeding the Diamond Y Springs system. Diamond Y Springs discharge a low to moderately saline Na-Ca-Cl-SO₄ type of water that is similar in composition to waters from the Rustler aquifer near Fort Stockton. The main processes affecting the water chemistry are: calcite, dolomite, halite, and gypsum dissolution and/or precipitation, and ion exchange between calcium, magnesium, and sodium (Boghici, 1997, 1999).

Stable and radiogenic isotope analyses in the Diamond Y Springs suggest that evaporation and water mixing processes are important controls on the spring water chemistry. Oxygen-18 and deuterium data indicate that Diamond Y waters are meteoric in origin; the data are distributed along an evaporation line according to spring discharge and pool size - the larger the discharge and the pool, the closer they resemble the main spring composi-

tion. The two main springs, Diamond Y main spring and Euphrasia spring, show tritium and ¹⁴C data indicative of a mixing of older and recent waters (Boghici, 1997, 1999). This occurs elsewhere in the study region (e.g., LaFave and Sharp, 1987; Uliana, 2000; Darling, 1997; Darling et al., 1995) and is supported by geochemical mass-balance modeling (Boghici, 1997). These are all consistent with the model of Diamond Y spring water being the product of mixing between Rustler Formation waters and recent local rain falling directly on the springs' pools. There is apparently discharge of older waters from the Rustler Formation, perhaps up along a fault trend, into Diamond Y Draw (Figures 1 and 3). Regional discharge, coupled with limited pumping from nearby wells to lower water tables, can explain the steadiness of these springs discharge. The spring flows then sink in the streambed and may reappear at downstream as seeps after undergoing various levels of evaporative concentration.

DISCUSSION AND CONCLUSIONS

All data and existing models support the hypothesis of extensive regional flow systems in Trans-Pecos Texas. Analyses of the fracture zones and the regional structural trends indicate that those trends influence, and probably control, the flow paths in this regional system. We also observe extensive structural features that connect recharge and discharge areas over great distances. Analyses of the fracture zones indicate that their orientations are controlled by pre-existing structural trends that have been reactivated over the history of this area. These demonstrate a pattern where ancient structural trends create the templates for fractures and karst patterns that control the development of a regional groundwater system. We hypothesize that this pattern may be repeated in carbonate systems in other semi-arid parts of the world. This has implications for our understanding of groundwater flow systems in regions that are usually dependent on groundwater for irrigation and municipal needs.

Regional flow systems connect individual groundwater basins. Fractures and subsequent karstification follow the structural trends and control the location of the major natural recharge and discharge areas (springs). At least 5 tectonic events and physical stratigraphic variability have led to a complex set of fracture domains. Mapping of these domains demonstrates how they have

controlled the development of the regional hydrogeologic system, including the karstification. Furthermore, the fractured (and karstic) systems inherent variability makes it difficult to predict the response of the groundwater systems to anthropogenic stresses and climatic variability. Detailed hydrogeologic assessments are required to utilize the region's groundwater resources and yet maintain critical environmental habitat. Natural tracer tests using strontium isotopes (⁸⁷Sr/⁸⁶Sr ratios) and fracture trace/intensity mapping are demonstrated to be promising assessment techniques of regional flow systems in this area or similar hydrogeological settings.

Diminishing spring flows or their cessation demonstrates the potential threats to remaining spring systems and their unique biota. Cessation is caused by a combination of climatic and regional factors. Increased groundwater extraction for irrigation and municipal use is the obvious cause, but other factors may also be important. These include the long-term pattern of increasing aridity in the region, possible alterations in the carbonate system permeability by either tectonic or sedimentation effects, localized pumping effects, reduction of recharge to the regional flow systems, and changes in the regional flow system boundaries, such as the beheading of the Wildhorse Flat - Toyah Basin flow sys-

tem. However, it seems clear in general that rates of springflow decline have increased with groundwater extractions for agricultural, municipal, and domestic use during the past century.

Municipalities (e.g., El Paso, Pecos, Fort Stockton, and Midland-Odessa) may need increased water resources. Irrigated agriculture is an economic mainstay for the area, and although the trends for irrigated agriculture are highly dependent upon economic conditions, the long-term needs for agricultural products are inferred to remain steady or increase. Increased utilization of groundwater will draw upon groundwater in storage both on a cyclical basis depending upon normal climatic variability and on a long term trend that could lead to overexploitation. Increased understanding of the regional

flow systems, including fracture controls, the nature of the recharge, and the flow paths, is needed to manage these resources. It may be possible to design pumping strategies that minimize the effects on natural springflows and yet meet projected demands. In addition, identification of key recharge areas and *a priori* analysis of fracture systems to identify fracture hydraulic domains may make it possible to maintain the springflows in the face of present or increased levels pumping (White et al., 1941; Mayer and Sharp, 1996; Uliana, 2000). We suggest that similar analyses and methodologies may prove of value in studies of spring systems in other areas of the southwestern United States and northern Mexico or other areas where regional flow systems exist in carbonate aquifers in semi-arid and arid zones.

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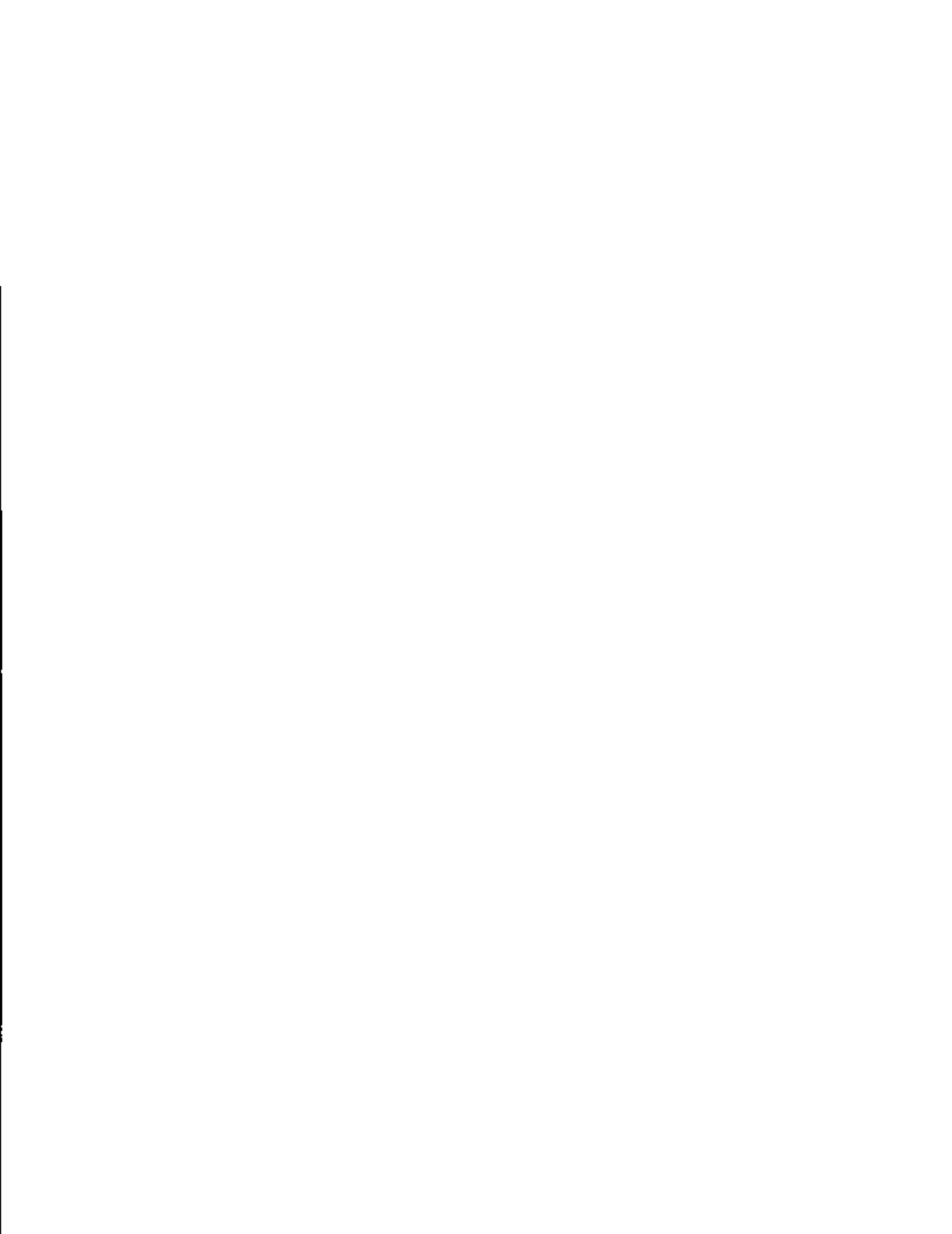
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CONTEMPORARY WATER SUPPLY IN WEST TEXAS AS AN EXAMPLE FOR THE NORTHERN CHIHUAHUAN DESERT

Kevin Urbanczyk

ABSTRACT

Water supply in the northern Chihuahuan Desert region occurs as both surface water and groundwater sources. The Rio Grande basin is the most significant of the surface water sources. This basin includes both the Río Conchos and the Pecos River systems. The Rio Grande Compact, the 1944 Treaty with Mexico regarding deliveries from the Río Conchos to the Rio Grande, and the Pecos River Program, governs water use in the basin. Groundwater resources in west Texas include several distinct aquifers. The Texas Water Development Board has categorized these aquifers as Major or Minor, based upon their size, geographic location, and geologic structure. Population growth predictions suggest large increases in the west Texas re-

gion, mostly in the El Paso area. These increases will place a significant strain on the available water supplies, and further use of both surface and groundwater sources will place a strain on the water available for wildlife. Within Texas, the Far West Texas Water Planning Group suggests several strategies for dealing with both the predicted population increase, and the decrease in available water due to drought and overuse of certain aquifers. These strategies include conversion of surface water appropriations from agricultural to municipal use, desalination, and interbasin transfer. Of these, the desalination and interbasin transfer of groundwater could significantly impact the ability of the groundwater to sustain wildlife populations.

INTRODUCTION

The northern Chihuahuan Desert encompasses parts of northern Mexico and the southwestern United States. As in all populated arid regions, water supply in this area is an essential issue for human habitation of the region. Complex legislation exists to control water use and, particularly in regions with expanding populations, overconsumption of water resources not only degrades the quality of life for humans, but significantly alters the natural ecosystems of this sensitive desert region.

This paper summarizes water supply in this arid region. Water "supply" is intimately associated with water "demand". This supply and demand concept is complex, and involves not only municipal water use, but also agricultural and industrial use. State, interstate and international government legislation is required to attempt to manage the use of this precious resource. Therefore, a summary of water supply necessarily

requires the inclusion of both a discussion of demand and of legislation governing water use. Water supplies in the northern Chihuahuan Desert occur as both surface and as groundwater. These two distinct sources of water occur in different areas, and are legislated in manners that also are quite different.

Figure 1 includes the Rio Grande drainage basin and major political boundaries. The Rio Grande drainage basin begins in southern Colorado, flows through New Mexico, enters Texas near El Paso, and continues to the Gulf of Mexico where it forms the boundary between the United States and Mexico. Significant tributaries to the Rio Grande in the northern Chihuahuan Desert region include the Río Conchos (confluence located at Presidio, Texas and Ojinaga, Mexico), and the Pecos River (confluence near Langtry, Texas).

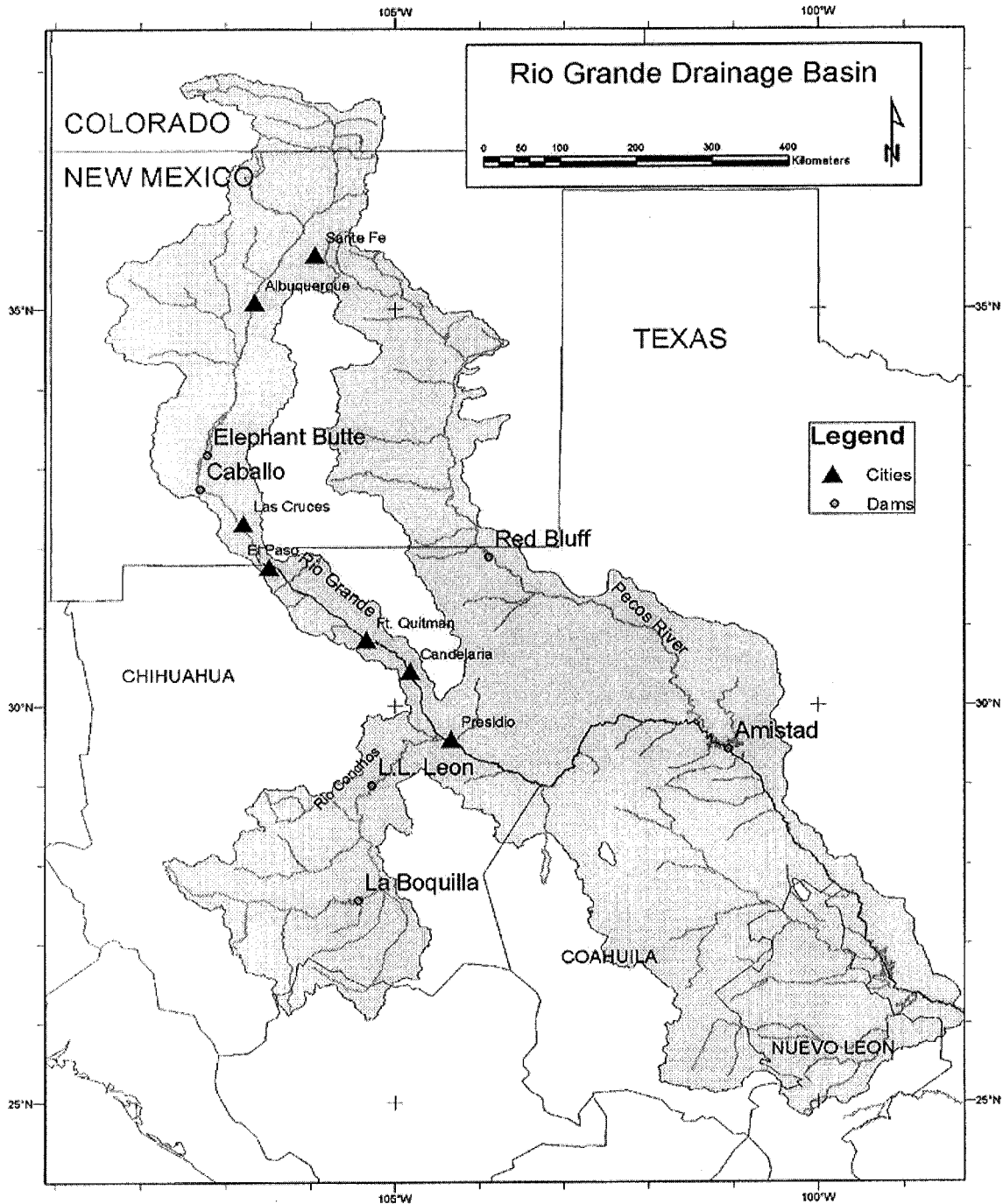


Figure 1. Rio Grande drainage basin with selected cities and dams. Basin boundaries and drainage patterns are from Texas Natural Resources Conservation Commission, and are available at <http://www.riogrande.org/programs/gis/gisdata.htm> (10/22/01).

Much of the content of this paper utilizes data available for the state of Texas. These data were produced in association with the development of the 1997 State Water Plan, and the development of the draft version of the 2002 State Water Plan. The data represent the most

complete compilation of water resources information for any section of the Chihuahuan Desert. Discussions of areas outside of Texas are included where appropriate and available.

WATER LAW

Water laws in Texas are similar to those in most other western states, but may differ from laws in Mexico. In Texas, water is classified as to where it physically occurs: percolating groundwater, underground streams, diffuse surface water, and streamflow (Wurbs et al., 1994). Of these, percolating groundwater and stream flow are the two significant water sources present in the northern Chihuahuan Desert. Groundwater has been historically governed by the “Right (or Rule) of Capture” doctrine. According to this doctrine, a landowner has the right to use or sell all of the water that can be captured from beneath a property (Wurbs et al., 1994). Stream flow is governed by the “Prior Appropriation”

doctrine. Surface water (in Texas) is publicly owned, and permits must be obtained to use surface water. The permitting, which is controlled by agencies such as the Texas Commission on Environmental Quality (TCEQ, formerly Texas Natural Resource Conservation Commission, TNRCC), is generally based upon who received the rights first (“first come, first serve”). Diffuse surface water is water such as return flow from an irrigated property. It is the property of the landowner until it reaches a watercourse. Underground streams are present in the northern Chihuahuan Desert. These feed water sources such as San Salomon springs in the Balmorhea area (Sharp et al., this volume).

SURFACE WATER RESOURCES

The Rio Grande basin encompasses approximately 180,000 square miles (466,000 square kilometers, Figure 1). It is best considered as two separate basins – an upper basin that ends at Presidio, Texas, and a lower basin that extends from there to the Gulf of Mexico. Additionally, the Pecos River and the Río Conchos can be considered part of the lower basin. Most of the flow in the upper basin is due to precipitation in southern Colorado and northern New Mexico (Wilson, 2000). This flow is impounded in a series of reservoirs. These include Elephant Butte (constructed in 1916; 2.11 million acre-feet (MAF) capacity) and Caballo (1938; 0.331 MAF) in southern New Mexico (Figure 1). As an example of evaporation rates, Elephant Butte and Caballo account for 85% of the 0.34 MAF/year that evaporate off of New Mexican Rio Grande reservoirs (Wilson, 2000). Dams in the Chihuahuan Desert region of the lower basin include La Boquilla (1916; 2.34 MAF) and Luis L. Leon (1968; 0.29 MAF) on the Río Conchos (Mexico), and Red Bluff (1921; 0.31 MAF) on the Pecos River (Texas).

The International Boundary and Water Commission (IBWC) has operated a series of gages in the Rio Grande basin since 1880 for some locations. Figure 2 shows the annual runoff past a series of these gages (IBWC; data from http://www.ibwc.state.gov/wad/rio_grande.htm). Several features are apparent upon observation of these data. First, there exists a general decline in discharge from Elephant Butte down river to Caballo, then to El Paso, and finally to Ft. Quitman. There have been periods of no flow in the Rio Grande at and

below Ft. Quitman, particularly during and after the drought of the 1950s. Second, the buffering effect of the installation of Elephant Butte dam on the discharge at El Paso is evident. Note the oscillation (higher high flows and lower low flows) evident in the early 1900s, which disappears after the construction of Elephant Butte Dam in 1916. Third, significant, yet declining, flow is provided by the Río Conchos near Presidio. Below the confluence of the two rivers, the Río Conchos provided 83% of the total downstream flow during the period 1961 to 1999, but only 55% during the period 1992 to 1999 (Brock et al., 2001). This demonstrates a considerable decline in the amount of water added to the main channel via the Río Conchos, a symptom of both Mexican water use and long-term drought. Fourth, from the mid-1990s on, the flow in both the Río Conchos and the Rio Grande has diminished to protracted low levels not seen since the drought of the 1950s. For a complete overview of the hydrology of the Rio Grande, see Schmidt, et al. (this volume).

Legislative action that governs appropriation of Rio Grande waters in the Chihuahuan Desert region begins with the 1902 (extended in 1905 to include western Texas) Reclamation Act (Littlefield, 2000). This act authorized the construction of Elephant Butte Dam and reservoir. Waters stored in the reservoir would be diverted to users through a system referred to as the Rio Grande Project. This 1905 law was the first interstate allocation of any river mandated by Congress. In 1906, an agreement was made between the U.S. and Mexico, which required that 60,000 acre-feet (AF) of water be allocated to Mexico

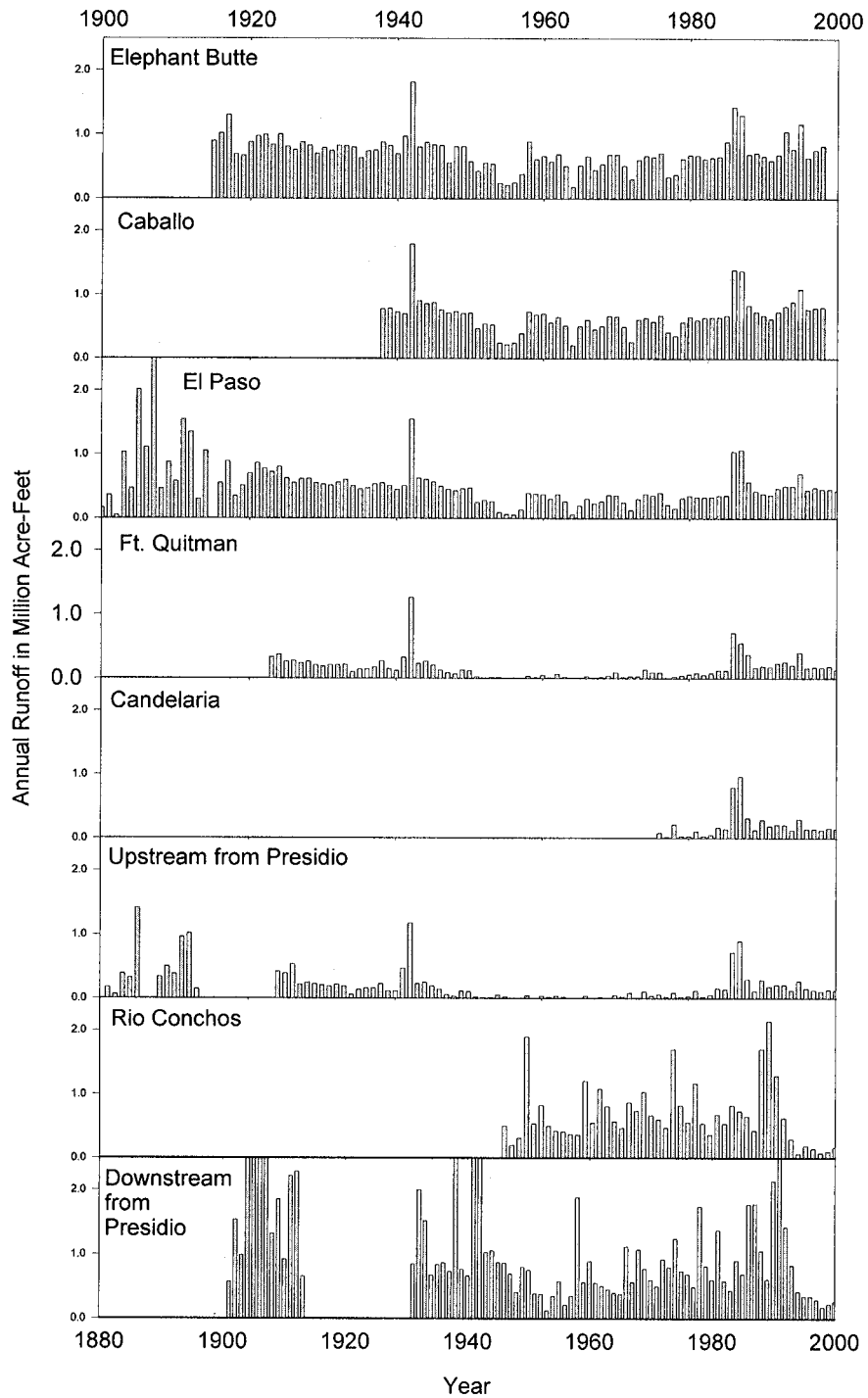


Figure 2. Annual runoff for selected sites along the Rio Grande in the northern Chihuahuan desert region (IBWC, 2002). The y-axis is fixed with a range of 0 to 2.5 MAF to aid in visual comparison between graphs. Note that for certain years the El Paso and downstream from Presidio graphs exceeded this value.

annually at a location upstream from Ciudad Juarez (Equitable distribution of waters of the Rio Grande Convention between the United States and Mexico, available at: http://www.ibwc.state.gov/FORAFFAI/1906_convention.HTM). Elephant Butte Dam was constructed in 1916, primarily for irrigation and flood control. Part of the appropriations from this project included enough water to irrigate 88,000 acres (35,600 ha) in New Mexico, and 67,000 acres (27,000 ha) in western Texas.

Appropriation of water between Colorado, New Mexico, and Texas is governed by the Rio Grande Compact of 1939 (Rio Grande Compact, reprinted in NMWRR, 2000). This compact defines the obligations of Colorado and New Mexico to deliver water ultimately to the Elephant Butte reservoir (from there to be distributed via the Rio Grande Project). The Rio Grande Compact requires minimum discharges be maintained at a series of gauging stations along the course of the upper Rio Grande. A complex set of equations was established to determine the amount of flow to be delivered from Colorado to New Mexico, and on to Elephant Butte. Total available water was therefore variable as discharge varied, but guaranteed flow to both New Mexico and Texas. Approximately 2.5 MAF of water are available for use within the Rio Grande Compact states annually. Most of this water is used to irrigate nearly 1,000,000 acres (405,000 ha) in the upper Rio Grande basin. Approximately 600,000 acres (243,000 ha) are in the San Luis valley of Colorado; additional irrigated acreage is located in the "middle" Rio Grande valley in New Mexico. Approximately 300,000 AF/year are used in the "middle" Rio Grande valley (Wilson, 2000), while 60 to 80% of flow at the Otowi gage (near Santa Fe) must be bypassed to Elephant Butte Reservoir due to Compact restrictions. Downstream from Elephant Butte, the Rio Grande Project (agricultural irrigation) and the delivery to Mexico (1906 Treaty) consume most (or all) of the remaining flow. Only in wet years is there any expected (or actual) flow downstream from Ft. Quitman. In fact, the average flow at Ft. Quitman is 140,000 AF/year, only 5% of the total water supply in the upper Rio Grande basin. The river is completely appropriated. New Mexico is currently in compliance with the Rio Grande Compact with regards to the delivery of water to Texas. It had accrued a total of 529,000 AF deficit to Texas after an all-time low of 19,000 AF stored in Elephant Butte in 1951 (Mutz, 2000). This deficit was erased by 1972, and New Mexico has been in compliance with the Rio Grande compact since then.

There continues to be diminished flow in the Rio Grande channel downstream from Ft. Quitman until the confluence with the Río Conchos (Figure 2). At this point, the Rio Grande once again becomes a perennial stream, with most of the water provided by the Río Conchos. Figure 2 demonstrates the impact the Río Conchos has on the main Rio Grande channel. Note the significantly different character of the annual runoff curve seen in the Rio Grande downstream from the Río Conchos confluence when compared to the Rio Grande upstream from Presidio. Also apparent on this Figure is the diminished flow provided by the Río Conchos since the mid 1990s. Appropriation of the Río Conchos, and delivery of water to Texas, is governed by the 1944 Treaty between Mexico and the United States (Treaty Between the United States of America and Mexico, available at <http://www.ibwc.state.gov/FORAFFAI/treaties.HTM>). In summary, Mexico is entitled to two thirds of the flow reaching the main channel of the Rio Grande through a series of rivers and streams, the largest of which is the Río Conchos. This is subject to the U.S. right to an average of 350,000 AF/year in cycles of five consecutive years. According to this agreement, Mexico is currently in a deficit situation (Kelly, 2001). As of October 1997, Mexico owed the U.S. 1.024 MAF, a figure that is twice the deficit incurred by Mexico during the drought of the 1950s. An additional 0.48 MAF deficit has been added as of early 2000 (Brock et al., 2001). According to the 1944 Treaty, Mexico is obligated to repay the water debt by October 2002. There does exist some question about this, though, because a provision exists in the treaty to alter the deliveries during "extraordinary drought". The exact definition of "extraordinary drought" is not made, and the drought of the mid-1990s might qualify.

Another water source in the western Texas region is the Pecos River. In its western Texas reach, flow in the Pecos River is controlled by releases from the Red Bluff Reservoir. The delivery of water from New Mexico is the primary control on storage in the Red Bluff Reservoir. The Pecos River Program (NMSEO, 1998) allocates Pecos River water between New Mexico and Texas. In 1988, the U.S. Supreme Court determined that New Mexico had underdelivered an average of 10,000 AF/year during the period 1950 to 1983. New Mexico agreed to pay \$14 million to Texas to eliminate this deficit. Today, average daily discharges along the Pecos River downstream from Red Bluff Reservoir vary from 4 to 15 cubic-feet per second (0.1 to 0.4 cubic-meters per second) (FWTRWPG, 2001).

GROUNDWATER RESOURCES

Groundwater is another major source of water for the northern Chihuahuan Desert region. Groundwater is inherently more difficult to study than surface water, particularly for estimating total resources. Aquifers are recharged by precipitation that infiltrates into the ground, by losing streams, by inflow from adjacent aquifers, and by irrigation return flow. Aquifer types in the general northern Chihuahuan Desert region include bolson type aquifers, alluvial aquifers, limestone aquifers, and igneous aquifers (Mace et al., 2001). The following discussion focuses primarily on the western Texas aquifers with information pertaining to the development of the 2002 State Water Plan for Texas. This plan will supersede the existing 1997 plan. The 2002 plan is the first to

be adopted since the passage of Senate Bill 1, which has allowed for more public participation in the production of regional water plans. The portion of Texas that includes the northern Chihuahuan Desert is located in Region E of the 2002 State Water Plan (Figure 3). This region includes the counties of El Paso, Hudspeth, Culberson, Presidio, Jeff Davis, Brewster, and Terrell. Associated with the water planning and the preparation of the 2002 State Water Plan has been research and data gathering by the Texas Water Development Board (TWDB). These data are used in the following summary of the most detailed assessment of water resources in the northern Chihuahuan Desert.

AQUIFER DETAILS

The TWDB has formally designated several aquifers in western Texas (TWDB, 1997; Figure 3). Note that in Figure 3, several of these aquifers overlap because the Figure includes both surface and subsurface spatial locations for the aquifers. These include the Hueco-Mesilla Bolson, Cenozoic Pecos Alluvium, and Edwards-Trinity (major aquifers), and the Bone-Spring Victorio Peak, Capitan Reef Complex, West Texas Bolsons, Igneous, Rustler, and Marathon (minor aquifers) (Mace et al., 2001). The Hueco-Mesilla Bolson and West Texas Bolson aquifers are located in sedimentary deposits associated with Basin and Range type extensional tectonics typical of the southwestern United States. This type of geologic activity produces linear mountain ranges separated by linear basins, which fill with sedimentary deposits as the mountain ranges rise over time. These linear basins are typically fertile sources of groundwater with recharge of water occurring primarily at the margins of the basins. Like other groundwater sources in this arid area, these aquifers are not recharged at rapid rates. The TWDB data compilation (TWDB, 1997) includes an estimate of "sustainable" water supply based upon precipitation and recharge rates for selected aquifers, and also includes an estimate of total useable (non-saline) water in storage for each of the aquifers. These figures for the Hueco-Mesilla Bolson aquifer are 0.024 MAF/year recharge, and 9 MAF storage, and 0.024 MAF/year recharge and 7 MAF storage in the West Texas Bolson aquifers. Recharge rates of 1% are estimated for the west Texas Bolson aquifers.

The Edwards-Trinity aquifer extends only partly into the Chihuahuan Desert region. This large aquifer system is located in Cretaceous limestone. It extends eastward into central Texas where it is connected to the Ogallala and the Edwards aquifers. It has approximately 145 MAF in storage, with an effective recharge rate of 0.776 MAF/yr.

The Cenozoic Pecos Alluvium aquifer is located in alluvial deposits of the Pecos River. This aquifer has an estimated recharge rate of 0.071 MAF/year, and 9.5 MAF in storage (available non-saline water). More than 200 feet (61 m) of water level declines have occurred in this aquifer in Reeves and Pecos counties. Groundwater that once contributed base flow to the Pecos River now flows in the subsurface to areas with heavy withdrawals.

The Bone Spring-Victorio Peak aquifer is located in joints, fractures, and cavities in Permian limestone beds. Recharge estimates are 0.09 MAF/year (annual recharge and irrigation return flow), and no total storage estimates are available. The Capitan Reef Complex aquifer is also located in Permian limestone, including the Capitan reef and reef talus. These limestone beds are commonly very porous (vuggy) and in extreme conditions may be cavernous (Carlsbad Caverns are located in this type of rock). Recharge estimates are 0.012 MAF/year, and total storage is estimated to be 0.385 MAF.

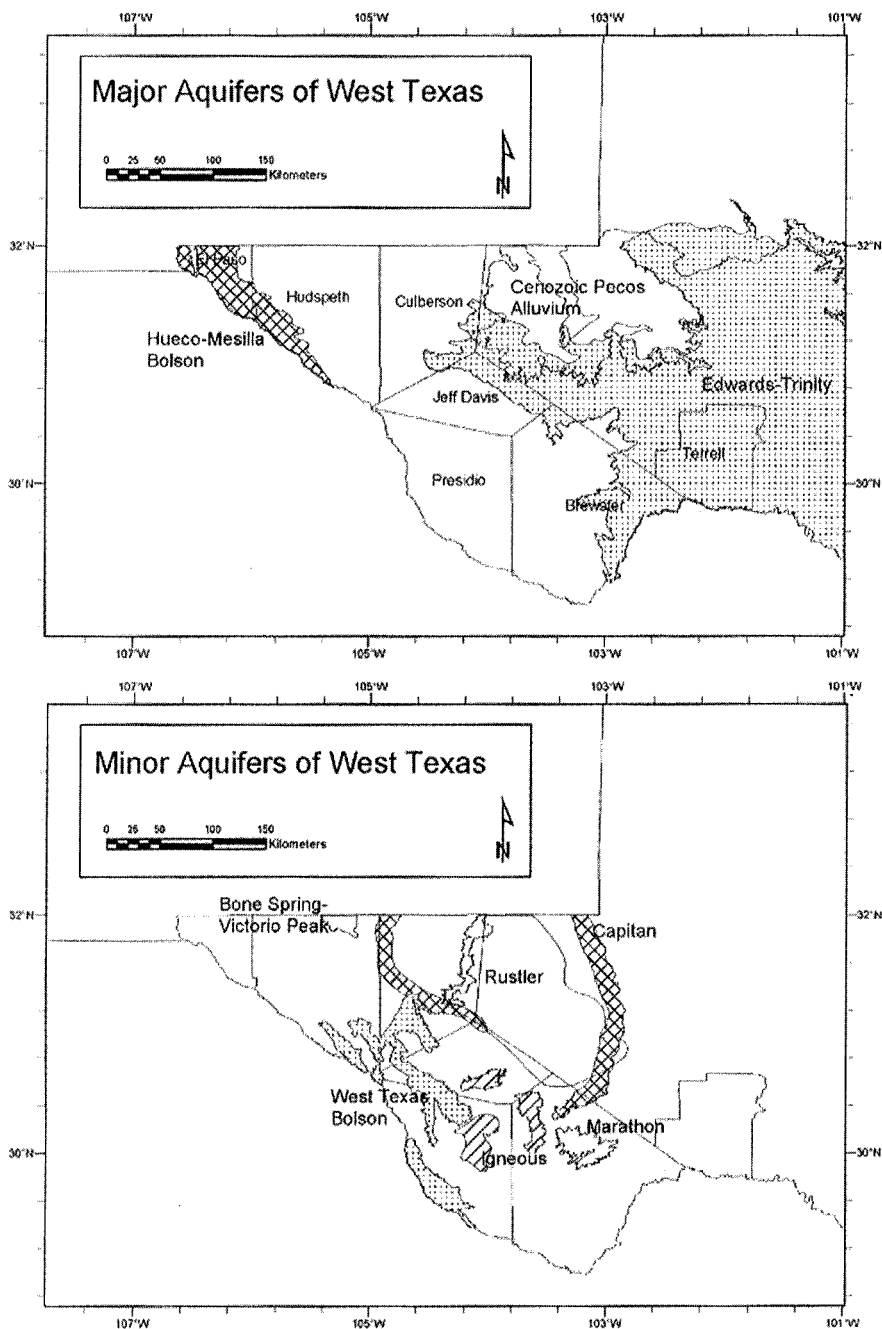


Figure 3. Major and minor aquifers of western Texas (TWDB, 2001b). The Region E counties are labeled in the top map.

The Igneous aquifers are located in Brewster, Presidio, and Jeff Davis counties. These aquifers are located in Tertiary (approximately 30 to 40 million years old) volcanic and volcanoclastic deposits, and associated recent alluvial sediments. They have estimated recharge rates of 0.014 MAF/year. The TWDB estimates total recharge rates in the aquifer system to be 2.5% of precipitation.

The Marathon aquifer is located in Paleozoic marine sediments. This aquifer has an estimated recharge

of 0.018 MAF/year, and a recharge rate of 2.5% of total precipitation.

The Rustler aquifer exists in up to 500 feet (152 m) of basinal limestone, dolomite, and evaporites representing the demise (drying up) of the Permian reef basin. The evaporite beds formed as the inland bay was cut-off from seawater circulation with the open ocean. Estimated recharge rates are 0.004 MAF/year. Water in this aquifer is not suitable for human consumption due to the high total dissolved solids (up to 6,000 mg/L).

SOCIAL IMPACTS AND SUPPLY VS. DEMAND PREDICTIONS

The population of Texas is expected to double in the next 50 years, from 21 million in 2000, to 40 million in 2050 (TWDB, 2001a). The Region E population is expected to increase from 800,000 to 1,587,097. The project population for El Paso county alone is 1,536,423, a 99% increase over the year 2000 census. By the year 2050, 38 percent of Texas' population will need to reduce demand or develop additional resources to meet projected demands during drought conditions (TWDB, 2001a). Agriculture, which is currently the largest water user in the state, will be surpassed by the combination of municipal and manufacturing demand. Impacts of this on western Texas include decreased irrigation due to depletion of groundwater resources and increase of groundwater use by urban centers such as El Paso. The TWDB recommends that the state Legislature establish protection of rural-community access to local water resources. They also recommend the use of groundwater models to evaluate the long-term sustainable levels of groundwater aquifers. Water demand in Texas is projected to increase from the current 17 MAF per year to 20 MAF in the year 2050. Region E demand is expected to increase from 0.509 to 0.586 MAF/year. Municipal demand is expected to increase by 67%, while irrigation demand is expected to decrease 12%. This decline will be due to more efficient irrigation systems and canal delivery systems, declining groundwater supplies, and the transfer of groundwater rights to municipal use as population increases. Current per capita water use throughout the state is 181 gallons/person/day (gpd), with a range of 275 gpd in Richardson and a low of 120 in Killeen (El Paso ranks near the bottom, using 144 gpd). Conservation efforts are expected to reduce the average per capita use to 159 gpd by 2050.

Total water supply projections (surface and ground, from existing sources) indicate an expected decrease of 18%, from 17.8 MAF/year in 2000 to 14.5 MAF/year in 2050 (TWDB, 2001b). Total groundwater *availability* is estimated by the Regional Planning groups to be 14.9 MAF/year. Total groundwater supplies (water accessible with existing infrastructure) are estimated to be 8.8 MAF/year in 2000, and are projected to decline 18% by 2050. Note that these "supply" estimates include the water available with existing infrastructure, and differ from the recharge and total storage estimates listed in the aquifer descriptions above. Statewide, groundwater constituted 50% of the total water supply in 2000 and is projected to provide the same in 2050. In Region E, groundwater constituted 79% of the total water supply in 2000, and is projected to supply 88% by the year 2050. These estimates again pertain to existing sources, and are skewed by the anticipated depletion of the usable portion of the Hueco-Mesilla bolson aquifer by the year 2030. The depletion of this major aquifer will create a critical need to find other sources to meet the projected growing demands for water. El Paso will clearly be unable to meet water demands by 2030 considering the existing supply. The Region E planning group recommends the following strategies to increase supply: 1) obtain additional surface water from conservation savings in irrigation; 2) purchase irrigation rights; 3) reuse; 4) desalinate; 5) purchase and use groundwater from outside of El Paso County. The impacts of groundwater transfers from rural counties will become a critical issue.

RELATED INFORMATION FROM NORTHERN MEXICO

Only about one quarter of 60 aquifers in Chihuahua have been studied in any detail (Kelly, 2001), and most water level measurements were suspended in 1990. The Mexican National Water Commission (Comision

Nacional de Aguas (CNA)) has identified several over-exploited aquifers which are listed in Table 1. Currently, only 1% of the wells have any type of metering (CNA, 1997).

Table 1. Major over-exploited aquifers in the Río Conchos basin (data from CNA, 1997).

Aquifer	Total Annual Pumping (MAF)	Total Annual Recharge (MAF)	% Over-Exploitation
Chihuahua-Sacramento	0.102	0.045	127%
Jimenez-Camargo	0.475	0.361	88%
Parral-Valle de Verano	0.026	0.021	21%
Tabaloapa-Aldama	0.054	0.045	19%

LONG-TERM PROGNOSIS

The long-term prognosis for the west Texas and northern Chihuahuan Desert region is difficult to determine. The projected population growth and anticipated depletion of groundwater reservoirs are critical water resource issues that need careful attention. Anthropogenic manipulation of surface and groundwater sources has and will further impact the ability of the water sources to sustain wildlife populations. This issue is compounded by the fact that we are in a significant drought. It is clear that we are using several of the groundwater sources at levels that are not sustainable. This has led policy makers to develop strategies to try to increase future municipal water supply. Some strategies proposed in west Texas, such as converting irrigation appropriations to municipal supply or increasing the amount of "reused" water, will not likely diminish the amount of available surface water for support of wildlife. However, desalination and interbasin transfer of groundwater could impact local surface water resources.

Sharp et al. (this volume) suggest that there are prolonged groundwater flow paths from locations such as the Wild Horse basin aquifer near Van Horn that likely feed spring systems such as San Salomon at Balmorhea. The export of water from the Wild Horse basin aquifer could, therefore, impact flow rates from these springs. Detailed studies such as these should be done to evaluate the impact of interbasin transfers of groundwater.

Careful attention must be given to compliance with respect to legislation and treaties governing deliveries of surface water between states and countries. It is important that we recognize how the effects of the current drought impacts Mexico's ability to deliver water via the Río Conchos. That deficit is definitely not the only surface water supply problem in west Texas. The over appropriation of the Rio Grande is at fault, also. There are no easy solutions to any of our water resource issues.

CONCLUSION

The information presented in this paper is representative of the water supply issues in the arid northern Chihuahuan Desert. Surface water resources are very limited, and are appropriated to the point of severe damage to natural ecosystems such as the Rio Grande. Many argue that we can adhere to legislation such as the Rio Grande Compact without the ecological destruction that currently occurs, and will likely occur in the future (Har-

ris, 2000). As population increases in this region, there will be a continuous shift to more dependence on groundwater rather than surface water to meet the growing demand. A necessary consideration will be the sustainable limits to groundwater withdrawals that will not completely mine these limited resources and will allow for the presence of both humans and wildlife.

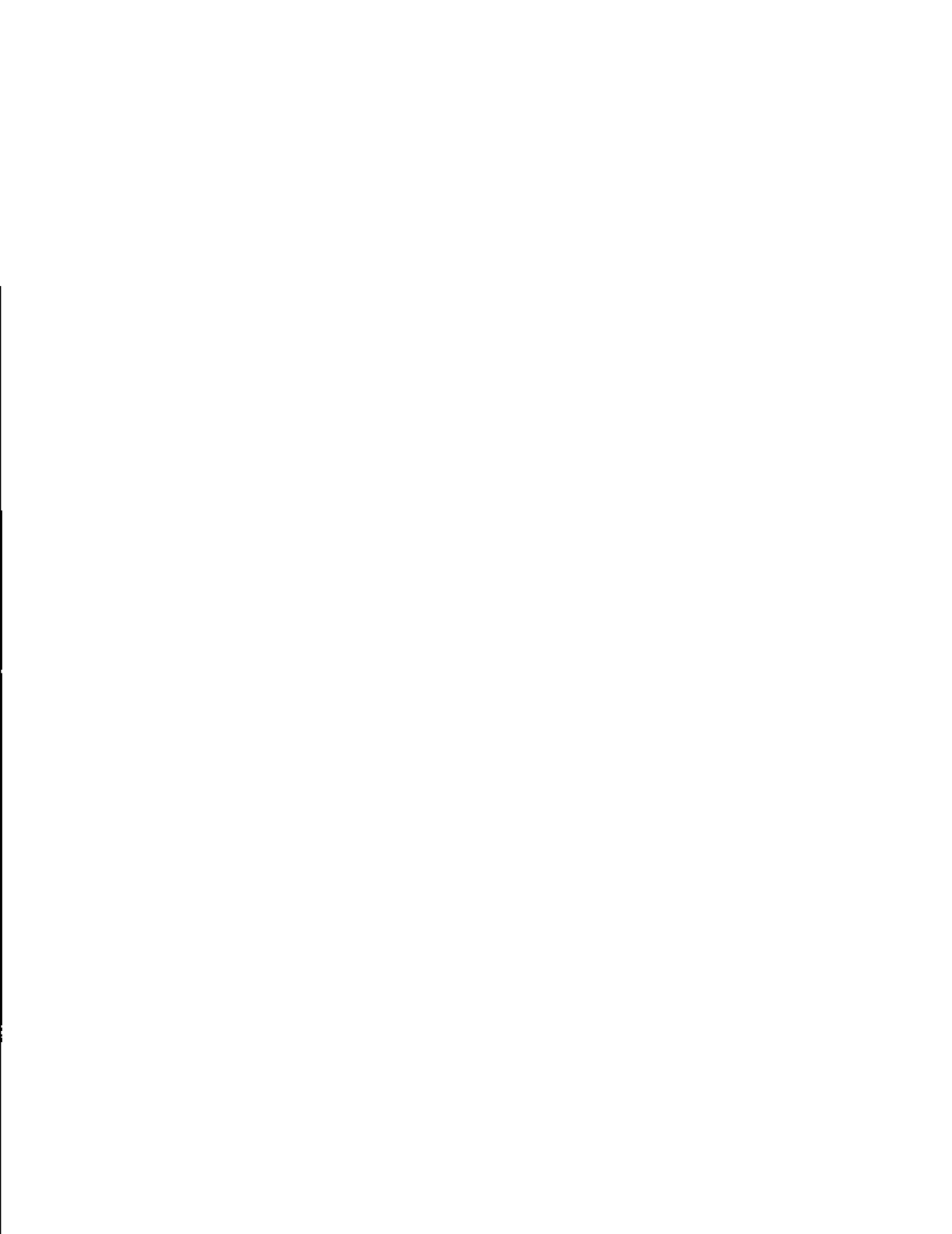
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HYDROLOGY AND GEOMORPHOLOGY OF THE RIO GRANDE AND IMPLICATIONS FOR RIVER REHABILITATION

John C. Schmidt, Benjamin L. Everitt, and Gigi A. Richard

ABSTRACT

The Rio Grande watershed includes a northern and southern branch that have very different hydrologic regimes. The natural flood regime of the northern branch is snowmelt driven, and that of the southern branch, the Río Conchos, is driven by summer rainfall. Downstream from the confluence of the two branches, near Presidio, Texas, the natural pattern of high and low flow was dominated by runoff from the Conchos basin between July and the following March prior to the construction of large dams. Dams and diversions greatly altered the natural hydrologic regime of both branches. The magnitude of the 2-year recurrence flood of the Rio Grande at El Paso, on the northern branch, declined by 76% after 1915. The magnitude of the 2-year recurrence flood downstream from Presidio was reduced by 49% after 1915.

Dams and diversions have also significantly altered the natural sediment flux, and significant geomorphic adjustments of the channel have resulted. The northern branch includes reaches where degradation

or aggradation has occurred during the past century. Reaches immediately downstream from dams have degraded beds and narrowed widths. Further downstream, the channel bed has aggraded, and the channel width has narrowed. Channelization and levee construction have occurred in some of these same river segments.

Restoration, defined as returning an ecosystem to a close approximation of its condition prior to disturbance, is impossible on the main stem of the Rio Grande because of current institutional demands on stream flow and the extent of alteration of the floodplain. Rehabilitation, defined as returning essential physical and ecological functions to a degraded ecosystem, is a more appropriate goal for the Rio Grande. In light of the diverse styles of twentieth-century channel adjustments that have occurred throughout the basin, different river segments must be assigned different rehabilitation goals.

INTRODUCTION

The Rio Grande has the second longest river course and had the sixth largest mean sediment discharge in North America before the continent was settled extensively by Europeans (Meade et al., 1990). Human activity has disrupted the natural flux of water and sediment. Large dams store floods for subsequent diversion, and these dams also trap sediment. The total volume of stream flow has been reduced, and the magnitude of floods in some parts of the Rio Grande have been reduced by more than 50%. Meade et al. (1990) estimated that annual sediment delivery to the Gulf of Mexico decreased from about 30×10^6 tonnes in 1700 to about 0.8×10^6 tonnes in 1980. These changes have caused significant adjustments of

the channel of the Rio Grande. Historically, the Rio Grande had a mobile bed and erodible banks, and the channel changed from year to year. Today's channel is smaller, more stable, changes less from year to year, and infrequently inundates its former floodplain.

The riverine ecosystem has adjusted to these changes in ways that do not benefit some native species. Inundation of the floodplain, which now occurs rarely in some segments, is necessary for recruitment in the riparian forest that lines the Rio Grande (Moles et al., 1998). Non-native salt cedar (*Tamarix* sp.) has widely colonized abandoned alluvial surfaces of the once-wider channel. The endangered Rio Grande sil-

very minnow (*Hybognathus amarus*) is adapted to the former wide shallow braided channel and associated habitats, and its population has declined greatly in response to channelization and diminished flows.

The purpose of this review paper is to describe hydrologic and geomorphic conditions of the river during the past century and to summarize changes in the water and sediment flux. We describe some of the geomorphic adjustments of the channel and its floodplain

that have occurred during the past century, emphasizing channel changes downstream from the large dams on the northern branch: Cochiti Dam and Elephant Butte and Caballo dams. These changes in hydrology, sediment transport, and physical characteristics of the channel and floodplain affect the aquatic and riparian ecosystem of the river. We conclude by commenting on the implications of these physical changes to development of a basin wide strategy for rehabilitating physical attributes and processes of the riverine ecosystem.

DESCRIPTION OF THE RIO GRANDE DRAINAGE BASIN

The hydrologic regime of the Rio Grande downstream from Presidio, Texas, results from the combined flow of northern and southern branches of the river (Figure 1). The drainage basin of the northern branch, called the Rio del Norte by Spanish explorers, comprises about two-thirds of the total watershed area upstream from Presidio. The flow of this branch, called the Rio Grande in the United States and the Rio Bravo in Mexico, is primarily contributed by snowmelt in the southern Rocky Mountains, and this branch had its annual peak flow in late spring, prior to the construction of dams. The Rio Conchos, whose headwaters are in the Sierra Madre Occidental, is the southern branch. Although the Rio Conchos basin is smaller than that of the northern branch, its mean annual runoff is much larger, and this branch has its maximum flows in late summer.

Several names are used to describe the different parts of the northern branch. The basin upstream from Elephant Butte Reservoir was referred to as the Upper Basin by Dortignac (1956) and the Northern Rio Grande by Graf (1994). Scurlock (1998) defined the segment between the Rio Chama and Elephant Butte Reservoir as the Middle Basin and many studies of this segment refer to it as the Middle Rio Grande, distinguishing it from the Upper Rio Grande that occurs upstream from the Rio Chama. For our purposes, we use the term northern branch when referring to the entire basin upstream from the Rio Conchos, and we refer to shorter river reaches by specific geographical names.

The river flows through a series of structural basins, where the alluvial valley is very wide, separated by intervening canyons where the valley is narrow. The occurrence of wide alluvial valleys and intervening narrow canyons is important in analyzing channel adjustment to the regulation of stream flow and sediment flux.

Rivers typically have lower gradients in wide alluvial valleys where they have large floodplains and meandering channels. Typically, channels narrow to greater extents in alluvial segments with flat gradients, and channel adjustments are less in narrow canyons (Grams and Schmidt, 2002).

The northern branch's headwaters are in the San Juan, Sangre de Cristo, and Jemez mountains of Colorado and New Mexico (Figure 1). The most upstream beginnings of stream flow occur near Stoney Pass in the San Juan Mountains. The Rio Grande leaves the San Juan Mountains near Del Norte, Colorado, and enters the San Luis Valley of south-central Colorado. This valley is a deep structural basin at the northern end of the Rio Grande Rift that is filled with more than 9,000 m of alluvium. The Rio Grande has a low gradient and has not significantly incised its channel through these sediments. Thus, the Rio Grande is easily diverted onto adjoining valley lands here, and irrigation is extensive.

South from the San Luis Valley, the Rio Grande enters a narrow canyon through the Taos Plateau – the Canon del Rio Grande. Further downstream are Espanola Basin, White Rock Canyon, and the Santo Domingo-Albuquerque-Belen basin. The large basins of central New Mexico have been aggrading for as much as 11,000 years (Sanchez and Baird, 1997), and the Rio Grande channel is not significantly incised into the sediments of the alluvial valley. Aggradation in these basins continues to the present. Two basins in southern New Mexico – Engle and Las Palomas Valleys – are partially inundated by Elephant Butte and Caballo reservoirs, respectively. The releases from these reservoirs are diverted for agriculture in the Mesilla and El Paso/Juarez valleys further downstream.

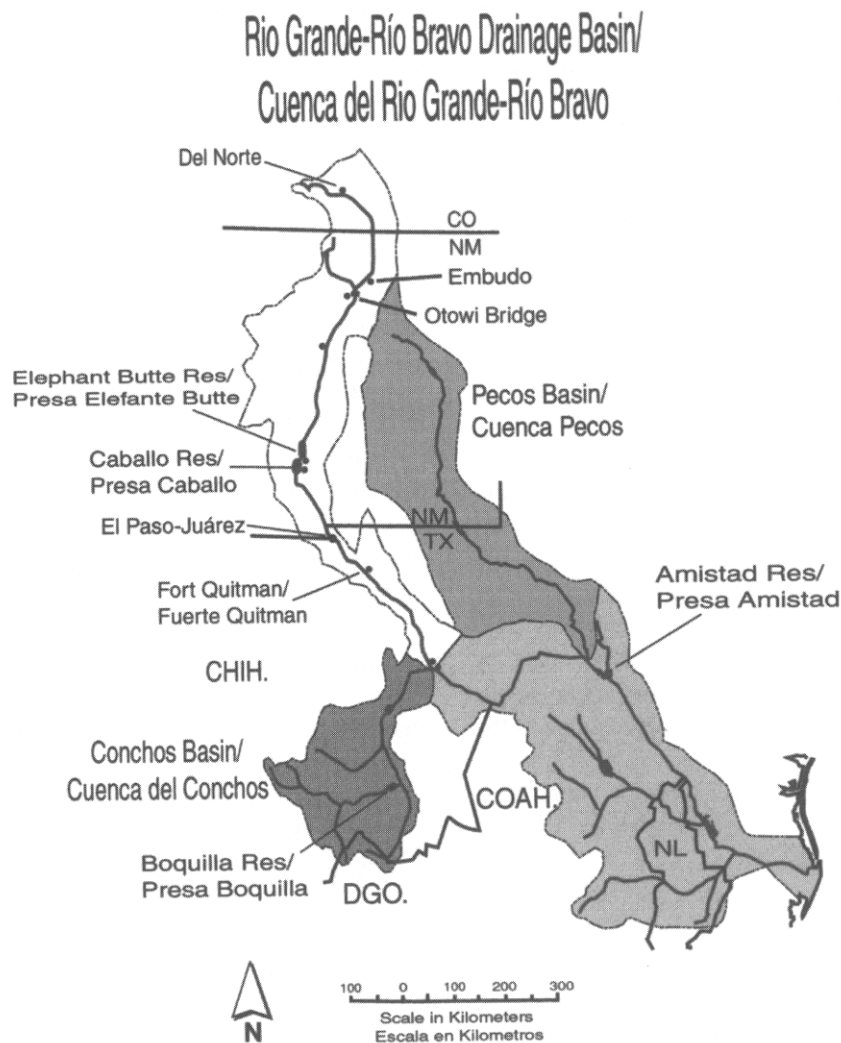


Figure 1. Map showing entire drainage basin of the Rio Grande/Río Bravo.

The El Paso/Juarez valley is about 136 km long, 16 km wide in places, and extends downstream to approximately Fort Quitman, Texas (Stotz, 2000). Downstream from Fort Quitman, the Upper Canyon segment includes 200 km where there are 11 different canyons and as many intervening alluvial valleys. The longest individual canyon is 14.7 km long, and canyon reaches comprise about 24% of the Upper Canyon segment upstream from Candelaria. The Upper Canyon segment also includes the Presidio Valley, which is about 120 km long between Candelaria and Presidio. The Presidio Valley is less than 5 km wide. The river is mostly channelized and leveed

here. The Rio Grande is joined by the Río Conchos near Presidio.

Downstream from the Río Conchos, the Rio Grande flows through alternating alluvial and confined reaches in the Big Bend section, including four narrow canyons that are popular for recreational boating – Colorado, Santa Elena, Mariscal, and Boquillas canyons (Aulbach and Gorski, 2000). The Lower Canyons extend to the headwaters of Amistad Reservoir (Aulbach and Butler, 1998).

Downstream from Amistad Reservoir, the Rio Grande exits its canyons and flows across the Gulf Coast piedmont. With the added contributions of the Pecos and Devils Rivers, it still occasionally lives up to its names "Grande" (Big) and "Bravo" (Wild). Peak flows from

occasional autumn hurricanes exceed 25,000 m³/s. Downstream from Laredo, Texas, the Rio Grande wanders across its delta plain of fine-grained alluvial deposits.

THE HISTORY OF WATER DEVELOPMENT

Agricultural use of the Rio Grande in New Mexico began in pre-history (Table 1). Pueblo peoples were utilizing ditch irrigation on a limited scale at the time of Spanish exploration in 1591 (Scurlock, 1998). Graf (1994) speculated that, "Diversion works on the main stream probably consisted of brush and boulder structures ... [that] probably washed away with each spring flood." Spanish and Mexican settlers in New Mexico expanded irrigation on floodplains and terraces of the Rio Grande, and the area of irrigated farming steadily increased in New Mexico until it reached a peak of 50,500 ha in 1880 (Sorenson and Linford, 1967, cited by Scurlock, 1998). Ditch irrigation began in the mid-1600s in the El Paso/Juarez Valley and direct diversions of the main channel in this valley were underway by at least the late 1700s (Stotz, 2000). Water was being diverted from the Río Conchos for use at the presidio in the Presidio Valley by 1750.

Of the 63 dams built in the northern branch watershed prior to 1916, 48 were in Colorado, and their purpose was to facilitate irrigation in the San Luis Valley. Between 1855 and 1893, 8 dams were built there whose cumulative reservoir storage was 4.08 x 10⁶ m³ (data from U. S. Army Corps of Engineers, 1996). Between 1894 and 1915, 55 more dams were built in the northern branch watershed, and the cumulative reservoir storage increased more than 100 times to about 486 x 10⁶ m³.

Depletions of stream flow caused by irrigation withdrawals have been substantial for more than a century. Kelley (1986) estimated that more than half the summer stream flow from central and northern New Mexico between 1890 and 1893 was consumed by irrigation. Kelley (1986) also estimated that 74% of the Rio Grande's stream flow was lost to seepage, evapotranspiration, and irrigation between the Mesilla Valley in southern New Mexico and Presidio during the same period. Without irrigation, Kelley (1986) estimated that losses would only have been about 35%. Between 1936 and 1953, the average annual depletion in the San Luis Valley was 9.9 x

10⁸ m³, and annual depletions ranged from about 6.2 x 10⁸ m³ in dry years to more than 12.3 x 10⁸ m³ in wet years. Depletions in central New Mexico were of a similar magnitude during this period (Thomas et al., 1963).

Elephant Butte Dam was completed in 1916, and had an initial capacity of about 2.93 x 10⁹ m³. The dam was built to control floods and ensure the delivery of irrigation water to southern New Mexico and to Mexico. At the time of completion, Elephant Butte Reservoir had a capacity of 2.5 times the mean annual discharge and was the largest reservoir in the world. Its construction increased the total reservoir storage in the basin by more than 6 times to 3,390 x 10⁶ m³ (Figure 2).

Small reservoirs, low head main stem diversion structures, levees, and channelization works were built throughout central New Mexico in the 1920s (Scurlock, 1998). These construction activities were directed by the Middle Rio Grande Conservancy District, organized in 1925. Diversion dams directed stream flow into extensive irrigation canals at Cochiti, Angostura, Isleta, and San Acacia. The construction of levees to prevent avulsions into surrounding agricultural lands along the river exacerbated the aggradation by confining sediment deposition to a smaller area (Scurlock, 1998; Sanchez and Baird, 1997). The construction of levees, begun in the 1920s, became a comprehensive channelization scheme that was completed in central New Mexico by the early 1960s (Graf, 1994).

El Vado Dam on the Rio Chama was completed in 1935 for flood control and irrigation supply. Caballo Dam, immediately downstream from Elephant Butte, was completed in 1938, and total basin wide reservoir storage increased to 4.37 x 10⁹ m³. Together, Elephant Butte and Caballo completely stored the annual snowmelt flood in every year between 1915 and 1941, and there were no flood releases downstream. The years 1941 and 1942 had unusually large runoff, however, and the dams and levees of that time were not able to control those floods.

Table 1. Dams and other structural modifications in the Rio Grande basin upstream from Amistad Reservoir.

Date	Event
1200-1850s	Pueblo, Spanish, and Mexican temporary diversion structures in the Rio Grande channel in New Mexico with gradual expansion of irrigated area in central New Mexico
1659	Founding of mission at Paso del Norte, temporary diversion and headgate constructed
1899	Cordoba Island cut-off, El Paso-Juarez
1916	Elephant Butte Dam completed
1925	Middle Rio Grande Conservancy District organized
1925-1935	Diversion dams at Cochiti, Angostura, Isleta, and San Acacia completed, 290 km of riverside drains and 260 km of interior drains constructed in irrigated fields of central New Mexico
1926	Salt cedar planted for erosion control in Rio Puerco basin
1933	Channelization through Mesilla Valley to El Paso
1935	El Vado Dam on the Rio Chama completed
1938	Caballo Dam completed
1938	Rectification and channelization, El Paso to Ft Quitman
1941 and 1942	Large floods cause 27 levee breaks near Albuquerque
1940s	Rio Puerco sediment control structures and revegetation
1950s and later	Channelization of the Middle Rio Grande
1950s?	Sediment control dams on tributary arroyos between Elephant Butte and Fort Quitman
1963	Abiquiu Dam completed
1967	Settlement of the "Chamizal" boundary dispute and construction of concrete-lined channel separating El Paso and Juarez
1969	Amistad Dam completed
1971	Heron Dam completed
1971	Transbasin diversion from the San Juan River
1973	Cochiti Dam completed

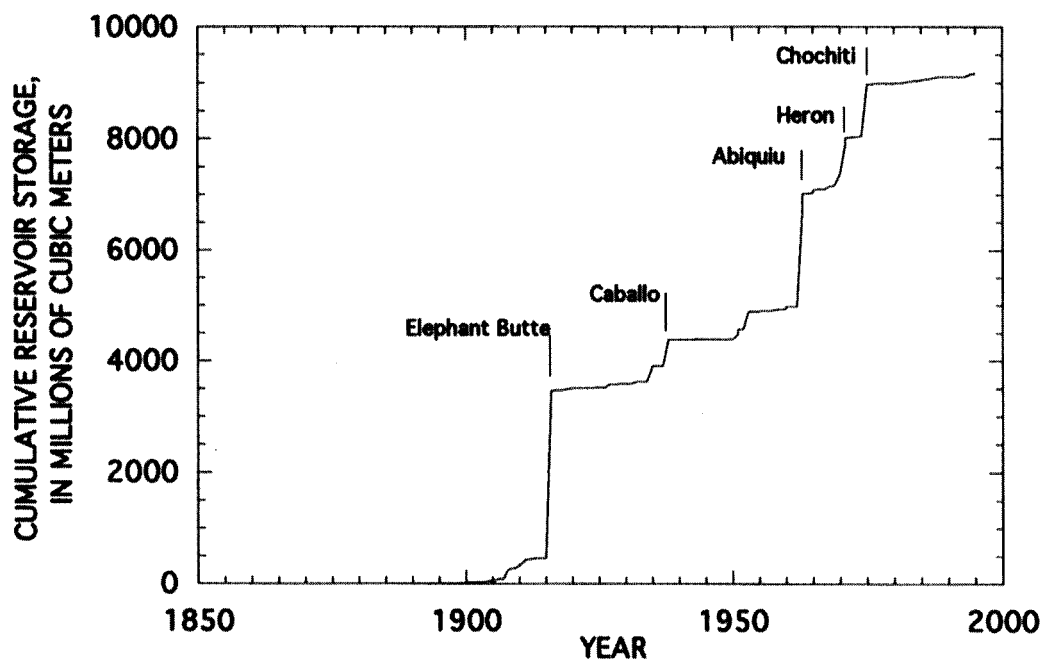


Figure 2. Graph showing time series of cumulative reservoir storage in the northern branch.

Abiquiu Dam was built on the Rio Chama in 1963 as part of the Colorado River Storage Project, and diversions from the San Juan River into the Chama basin began in 1971. Today, Abiquiu is the second largest dam in the northern branch watershed.

Cochiti Dam on the Rio Grande, located 65 km upstream from Albuquerque, was completed in 1973. It provides the largest flood control storage volume on the northern part of the main stem (Bullard and Lane, 1993). The dam was completed in November 1973 for flood control and sediment detention (U.S. Army Corps of Engineers, 1978) and traps virtually the entire sediment load from upstream (Dewey et al., 1979).

Large dams have also been constructed on the Río Conchos, creating La Boquilla Reservoir in 1913, Francesco I. Madera Reservoir in 1947, and Luis L. Leon Reservoir in 1967. The largest reservoirs in the Rio Grande basin are located downstream from Presidio: Falcon (completed in 1954; $3.18 \times 10^9 \text{ m}^3$) and Amistad (completed in 1969; $5.13 \times 10^9 \text{ m}^3$). The cumulative size of Amistad and Falcon reservoirs is greater than the total storage of all the reservoirs of the northern branch, illustrating the substantially greater stream flow that is regulated in the downstream parts of the Rio Grande/Río Bravo.

TREATIES

The international position of the Rio Grande has played a significant role in its physical and hydrologic history, as well as its cultural history (Mueller, 1975). Under the Treaty of 1848, the segment between El Paso and the Gulf of Mexico was made the boundary between the two countries. Mapping of the river boundary was completed in 1852 (Emory, 1857). The active nature of the river was not anticipated, and within 30 years parts of the river had wandered kilometers from its 1852 course and dozens of oxbows were abandoned, making redefinition of the boundary necessary. The Treaty of 1884 included specific language providing for a moveable boundary, following the natural migration of the channel by erosion and accretion, but remaining fixed in the abandoned channel in the event of avulsion. The treaty also provided for additional mapping of channel changes, and restricted artificial modification of the channel.

The Treaty of 1970 provided for the first complete mapping of the 2000-km river boundary since 1852. The treaty strengthened restrictions against artificial modification to include levees on the flood plain that might raise flood heights on the opposite bank.

The Water Treaty of 1906 apportioned the flow of the northern branch, and provided for storage and delivery of Mexico's allotment via the Rio Grande Project. The 1944 Water Treaty allocated the water of the Rio Grande downstream from Presidio and gave the International Boundary and Water Commission authority to oversee measurement and distribution of stream flow. The treaty provided for the construction of international storage reservoirs. Reflecting the wartime emphasis on agriculture and industry, the treaty established the following priority for use of stream flow: domestic and municipal uses, agricultural and stock-raising, hydroelectric power generation, other industrial uses, navigation, fishing and hunting, and other beneficial uses.

HYDROLOGY OF THE BASIN PRIOR TO 1915

The records of floods and droughts on the northern branch are preserved in the journals and notes of explorers and residents of the basin. Scurlock (1998) determined that there were at least 50 major floods exceeding $280 \text{ m}^3/\text{s}$ in New Mexico between 1849 and 1942 and 51 floods in the El Paso/Juarez Valley since 1846. Twice as many known floods occurred in the 1800s than in the 1600s or 1700s. Scurlock (1998) and Stotz (2000) suggested that environmental degradation may

have contributed to the increase in flood frequency in the 1800s, but Graf (1994) suggested that regional climate change was a more likely cause. The largest flood occurred in 1828 and had an estimated discharge of about $2,830 \text{ m}^3/\text{s}$. During this flood, the entire Rio Grande valley was inundated from Albuquerque to at least El Paso. Other very large floods occurred in 1872 and 1884.

The gauged flow of the Rio Grande prior to 1915 reflected the impacts of irrigation withdrawal in the San Luis Valley and central New Mexico. The northern branch flooded in late spring, with a secondary peak in summer (Scurlock, 1998). The magnitude and average duration of the spring snowmelt flood increased in the downstream direction between the San Juan Mountains and central New Mexico, as reflected in the difference between measurements near Del Norte, at Embudo, and at Otowi Bridge (Figure 3). Between central New Mexico and El Paso, the magnitude of the snow melt flood did not increase, however, because there are no other large tributaries that drain high mountain ranges with significant annual snow fall. Thus, the magnitude of the spring snowmelt flood at Otowi Bridge was nearly the same magnitude as at El Paso (Table 2).

Prior to 1915, the reach between El Paso and Presidio was a losing stream due to seepage losses, evapotranspiration, and irrigation diversions (Kelley, 1986). The entire flow was sometimes diverted at El Paso, resulting in occasional dewatering of the river downstream (Everitt, 1993). The magnitude of the 2-year recurrence flood, prior to 1915, decreased from 209 to 122 m³/s between El Paso and the Río Conchos (Table 3). In those years when the annual peak flow at El Paso was less than 100 m³/s, no snowmelt flood peak reached the Río Conchos. In years of greater snowmelt runoff, the magnitude of the peak flow at the Río Conchos was never more than 90% of that measured at El Paso, and typically occurred 7 to 10 days after the peak had passed El Paso. The only times when stream flows at Presidio were significantly larger than at El Paso were in the late summer and early fall when flood flows were triggered by rainfall in the downstream parts of the basin.

Table 2. Summary of hydraulic characteristics of the Rio Grande at selected gauging stations in late 1800s and early 1900s, before completion of Elephant Butte Dam.

Gauging station location and period of record	Median annual maximum mean daily discharge, in cubic meters per second	Median date of the annual maximum mean daily discharge	Mean annual discharge, in cubic meters per second ¹	Number of days whose median discharge exceeded twice the mean annual discharge
near Del Norte (1/1/1890-5/31/1890; 7/1/1890-9/30-1896; 1/1/1904-12/31/1906; 1/1/1908-9/30/1915)	107.3	June 13	27.2	60
at Embudo (1/1/1889-3/31/1904; 9/1/1912-9/30-1915)	121.7	June 5	29.5	62
at Otowi Bridge (2/1/1895-12/31/1906)	146.9	June 4	37.3	61
at El Paso (5/10/1889-6/30/1893; 1/1/1897-12/31/1897; 2/1/1898-9/30/1915)	147.0	June 18	35.1	66
above Río Conchos, near Presidio (1/23/1900-1/31/1900; 2/23/1900-2/28/1900; 3/23/1900-3/31/1914)	77.0	May 26	22.4	63
below Río Conchos, near Presidio (5/1/1900-5/31/1914)	219.0	September 6	72.1	50
at Langtry (5/1/1900 – 9/30/1913)	694	August 14	70	—

¹ computed as the mean of all days when measurements were made

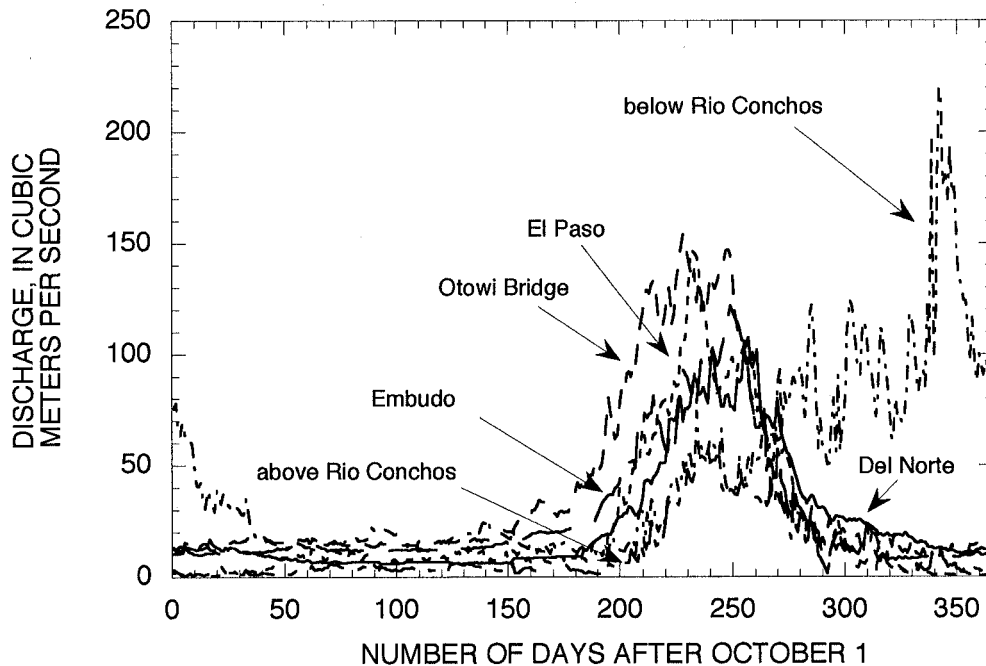


Figure 3. Graph showing median hydrographs of mean daily discharge of six gauging stations of the Rio Grande for varying periods in the late 1800s and early 1900s. See Table 2 for periods of record for each station.

Table 3. Magnitude of floods of different recurrences, upstream and downstream from the Río Conchos.

	Discharge, in cubic meters per second, of the annual maximum mean daily discharge, for the indicated period at the indicated location			
	1.25 yr	2 yr	5 yr	10 yr
1898-1916				
at El Paso	98	209	378	484
above Río Conchos, near Presidio	52	122	244	330
below Río Conchos, near Presidio	217	567	1160	1545
1916-1996				
at El Paso	32	51	100	124
above Río Conchos, near Presidio	15	34	70	99
below Río Conchos, near Presidio	126	288	661	1023

The natural hydrology of the Río Grande changed dramatically downstream from the Río Conchos (Figure 3). The Río Conchos' hydrology is entirely determined by rainfall, which is greatest in late summer and early fall in the Sierra Madre Occidental. This watershed yields the bulk of its natural stream flow between July and the following March (Table 4). Prior to 1915, the magni-

tude of peak flows downstream from the Río Conchos was approximately four times what they were upstream (Table 3). During September, when the Río Conchos reached its annual maximum discharge, approximately 93% of the lower Río Grande's total monthly flow came from the Río Conchos.

Table 4. Mean monthly discharge of the Rio Grande/Río Bravo upstream and downstream from the Río Conchos, near Presidio, 1901-1913.

Month	Mean monthly discharge above Río Conchos, in cubic meters per second	Mean monthly discharge below Río Conchos, in cubic meters per second	Percentage of mean monthly discharge of the lower Rio Grande/Río Bravo that originated in the northern branch
October	218	1002	22
November	125	592	21
December	117	537	22
January	101	298	34
February	95	362	26
March	139	297	47
April	233	291	80
May	723	795	91
June	1013	1273	80
July	543	1478	37
August	205	1828	11
September	207	2797	7

CHANNEL CHARACTERISTICS IN THE 1800s AND EARLY 1900s

The northern branch was an aggrading stream whose braided channel was constantly shifting (Graf, 1994). Large loads of sandy sediment and widely fluctuating flows caused the channel to be very wide and relatively shallow. As described in the El Paso/Juarez Valley by Major O.H. Ernst of the Army Engineer Corps in 1896 (cited by U.S. Department of State, 1903), "The size and character of the [Rio Grande] are ever varying, and its requirements as to form and dimension of bed vary equally. The river's work of altering its bed to suit the necessities of the moment is never ending." Channel change data for the part of the Rio Grande that is the international boundary demonstrate that the channel was very active and migrated rapidly across its sandy flood plain by both lateral erosion and avulsion (Mueller, 1975). Channel avulsions that were typically meander cutoffs during floods were most common in the wide alluvial valleys.

In central New Mexico, the channel was generally straight with numerous braided channels. In the El Paso area, the channel had a meandering course at flood stage, had a braided channel at low flows, and changed course

frequently. There is some evidence that the Rio Grande near El Paso had a narrow sinuous channel in early historic times, suggesting that the wide shallow channel of the late 1800s was perhaps the result of a "metamorphosis" (Schumm, 1969) resulting from the flood of 1828.

In central New Mexico in 1944, the Rio Grande at base flow was described by Rittenhouse (1944, p.150) as a ... "winding, elongated sand flat, averaging about 200-300 yards in width. One or more small low-water channels meander over the sand flat, re-working the deposits in it. At high stages the entire sand flat, as well as the adjacent floodway area beyond the low banks, is under water. Between large floods the width of the sand flat is decreased by growth of cottonwoods and salt cedars. These may be removed or the entire channel shifted during high flows."

Rittenhouse (1944) also noted that the floodway was nearly 1 km wide. Lateral movements of the Rio Grande downstream from Cochiti Dam between 1918 and 1935 averaged 20 to 35 m/year (Richard, 2001).

HYDROLOGIC CHANGES IN THE BASIN

Changes to the hydrology of the Rio Grande since 1915 have been profound. Peak discharges declined upstream and downstream from Elephant Butte Dam. Changes upstream from the dam are probably due to regional climate change as well as changing patterns of irrigation diversion. These changes greatly diminished the magnitude and duration of the annual peak flood and changed the season in which these floods occur. The net effect of all changes has been to make the magnitude of the annual floods more similar throughout the northern branch (Figure 4). In fact, the average flood at Del Norte is now larger, on average, than the magnitude of floods at El Paso.

Ainsworth and Brown (1933) summarized the effect of the recently-constructed Elephant Butte Dam on the downstream water and sediment flux: "Elephant Butte Dam and Reservoir have retained the entire flow of the Rio Grande entering the reservoir during the period of operation, 1916 to date [1932]. Release of water has been entirely under control and predicated on irrigation

demand [and] exceeds 2,000 second-feet for only short intervals. Practically all the silt (20,000 acre-feet annually) entering the reservoir from upper river sources has been retained above the dam."

These changes are illustrated by the median hydrograph for the period 1924 to 1940 for the reach between El Paso and Presidio (Figure 5). The well-delineated spring snowmelt peak was eliminated, and moderate flows at El Paso extended between April and September. These stable flows facilitated efficient agricultural water withdrawal in the El Paso/Juarez valley, as is evident in the difference between stream flow measured at El Paso and at Fort Quitman. These changes caused the magnitude of annual floods to be reduced by about 65 to 75% for the flows in the El Paso/Juarez valley (Table 3). In contrast, the magnitude of flood peaks downstream from the Río Conchos only decreased by between 33 and 49%, because the magnitude of flood control provided by reservoirs in the Conchos basin is not nearly as great as in the northern branch.

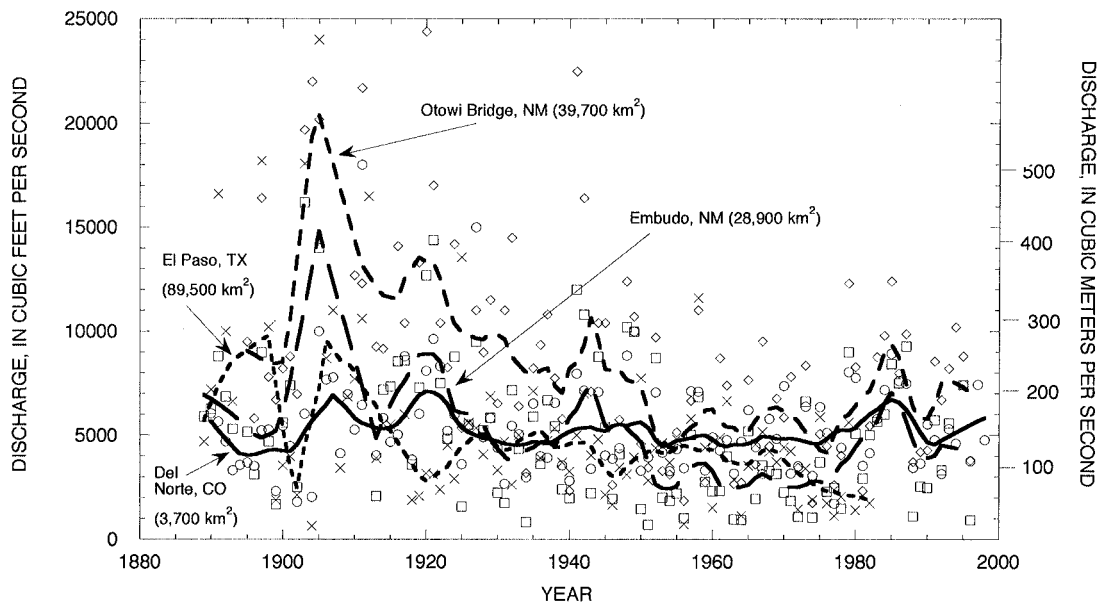


Figure 4. Graph showing the time series of annual maximum mean daily discharge at four gauging stations along the northern branch.

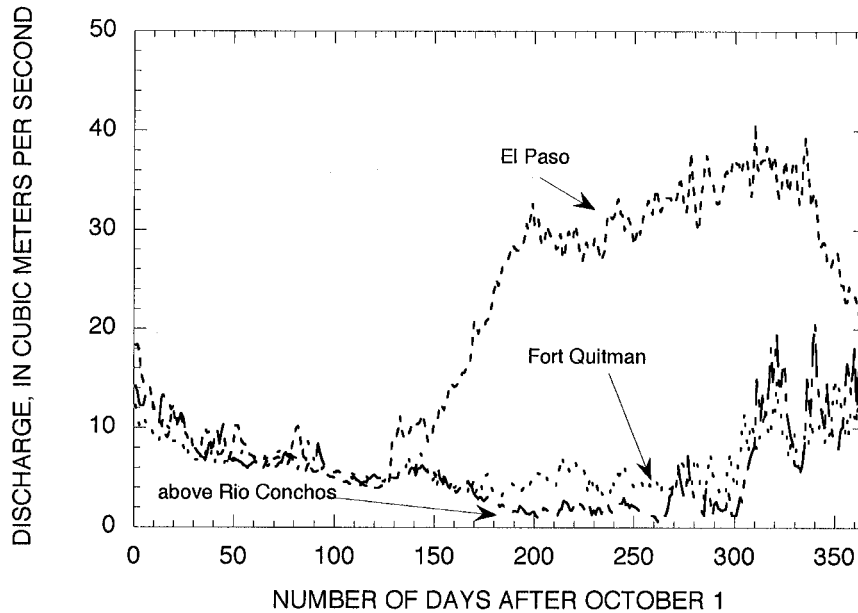


Figure 5. Graph showing median hydrographs of mean daily discharge for the Rio Grande for the period 1924-1940 at El Paso, at Fort Quitman, and above Rio Conchos, near Presidio.

Cochiti Dam affects the annual floods and the sediment input to the reach directly downstream. Cochiti Dam operations reduce those few floods exceeding 142 m³/s, resulting in a 38% decrease in annual floods from the pre-dam (1895 to 1973) to post-dam (1974 to 1995) period at the Cochiti gage, just downstream from the dam. Further downstream at the Albuquerque gage, the impact is diminished and the annual flood was only reduced by 4% following completion of the dam. The duration of peak flows increased 60 to 130% between the same time periods (Richard, 2001). Completion of Cochiti Dam resulted in a 99% reduction in sediment

concentration flowing into the channel downstream. Upstream from the dam, at the Otowi Bridge, the suspended-sediment concentration also declined around this time, and thus some aspect of reduced sediment concentrations may be due to regional climate and land use change. Suspended-sediment transport increases downstream from the dam due to re-supply of fine sediment from tributaries and/or erosion of the bed and banks. As a result, the post-dam reduction in annual mean suspended sediment concentration at the Albuquerque gage is 78% (Richard, 2001).

RESULTING CHANNEL CHANGES

Today, the northern branch between Cochiti Dam and Presidio can be divided into two long segments – one segment affected by the existence and operations of Cochiti Dam and the other affected by the existence and operations of Elephant Butte and Caballo dams. In the two segments, the reach nearest the dam has experienced bed degradation and coarsening of bed material. Further downstream in each segment, the channel has aggraded in reaches where the combined influx of sediment from tributaries exceeds the diminished transport

capacity of the river. The degrading reach downstream from Cochiti Dam probably extends to San Acacia, although smaller diversion dams at Angostura, Isleta, and San Acacia complicate this longitudinal pattern. The aggrading reach extends from there to the head of Elephant Butte Reservoir. The degrading reach downstream from Caballo Dam once extended to the Mesilla Valley, but channelization has obliterated this evidence. The channel has significantly aggraded downstream from El Paso, but a natural channel only exists downstream from Fort

Quitman. Within each segment, reaches have received different cultural treatments, in terms of direct manipulation to the channel and floodplain (Table 5).

The description of channel change between Cochiti Dam and Amistad Reservoir is based on an unusually comprehensive set of geomorphic data. The combination of severe flooding and sedimentation between Cochiti and Elephant Butte, along with irrigation needs in the

middle Rio Grande valley in the early 1900s prompted state and federal agencies to begin intensive surveys of the river. These surveys include cross-section surveys beginning in 1918, bed material sampling beginning in the 1930s, suspended sediment sampling beginning in the 1940s, and aerial photography (Leon et al., 1999). Changes along the international boundary are monitored by the International Boundary and Water Commission.

Table 5. Summary of cultural impacts to the Rio Grande between Cochiti Dam and Amistad Reservoir.

Reach		Flood Regime		Cultural Treatment		
From	To	Pre-1915	Post-1915	Channelization	Regulation	Depletion
Cochiti	Elephant Butte	spring	spring	moderate??	moderate	minor
Caballo	El Paso	spring	summer	moderate	extreme	moderate
El Paso	Fort Quitman	spring	summer	extreme	extreme	extreme
Fort Quitman	Candelaria	spring	summer	minor	extreme	extreme
Candelaria	Presidio	spring	summer	moderate?	extreme	extreme
Presidio	Amistad Reservoir	summer	summer	none	moderate	moderate

CHANNEL CHANGES DOWNSTREAM FROM ELEPHANT BUTTE AND CABALLO DAMS

Completion of Elephant Butte Dam caused the channel to degrade immediately downstream from the dam. Further downstream, the channel began to shrink in size in the El Paso/Juarez valley, because the low-gradient channel could not transport this delivered load, nor the load sluiced to the channel from irrigation channels or delivered naturally from ephemeral tributaries. Ainsworth and Brown (1933) reported that: "Silt carried in suspension past El Paso now varies from 0.03 percent to 1.5 percent by volume, depending upon the ratio of arroyo runoff to reservoir releases. But by far the greater part of the material transported is sand traveling probably as bottom load. This is either scoured from the riverbed or from the arroyo fans which are annually replenished by run-off from the summer rains. The controlled flow in the river is successively depleted for irrigation use at the various diversion points along its course, at each of which, through operation of skimming weirs and sand sluiceways, a great part of the sand is returned to the riverbed. The ordinary flow of the river by El Paso is not capable of transporting the load of sand and silt annually brought down the river from above."

"Peak flows at El Paso since Elephant Butte Dam, as a result of the above factors, are of annual occurrence, and while usually under 4,000 second-feet, have

amounted to 13,500 second-feet. The peak of these floods is sharp, lasting but a few hours so that the total acre-footage passed is low. The short duration of these summer floods precludes, as to the valley below El Paso, any lasting scouring action or long distance transportation of the accumulated deposits. Their action is more to carry the sand scoured from the bed over banks onto the flood plain, which is thus being constantly elevated. A general lowering of the river bed above El Paso has taken place [while] a general filling of the riverbed below El Paso has taken place. Narrowing of the normal channel has progressively occurred both above and below El Paso. This effect is most marked below El Paso where the normal channel has only about one third its former width."

"River gradients have been but little disturbed except where cut-offs have been made and in the immediate reaches above diversion dams or above plugs (of sediment) deposited by side flow and except for the reach of river immediately below the International Dam where filling has resulted in an increase in gradient from 2.45 feet per mile in 1917 to 3.00 feet per mile in 1932. However, decreasing gradients due to increasing river lengths are apparent below El Paso where the natural length has been undisturbed by cut-offs."

“River length above El Paso has apparently been slightly shortened by natural processes and reduced about five miles by artificial cut-offs. River length below El Paso has been lengthened by nearly 20 percent (compared to 1907) and by about 4 percent (compared to 1917) in those reaches where neither cut-offs have been made or avulsions occurred. River length above and below El Paso, when compared to valley axial length, has a ratio of 1.21:1 and 1.91:1, respectively.”

“The processes of the adjustment of the bed of the river to the new conditions of flow are not complete”

Channel shrinking on the Rio Grande provided the first clear evidence that large main stem dams in the western United States do not necessarily provide downstream flood control, and under certain conditions may actually increase flood risk due to diminished channel capacity. It spawned a short-lived discussion in the engineering literature of the 1920s and 1930s regarding the long-term effect of structural methods of flood control (Lawson, 1925; Stevens, 1938). The problem, from the point of view of water supply and flood control, was summarized in the Joint Report of the Consulting Engineers, International Boundary Commission, on rectification of the Rio Grande (IBWC, 1933): “Notwithstanding the fact that the present total amount of sediment annually carried through this valley by the Rio Grande is only a very small percentage of that carried previous to the construction of the Elephant Butte Dam, the absence of the former large scouring floods has resulted in the silting up of the river channel to a point where rainfall discharges from arroyos entering the river between Elephant Butte and El Paso-Juarez menace the improved and developed properties of both cities and valley lands. Only large floods of destructive proportions are capable of eroding accumulations of sediment as they now occur in the meandering channel.”

This report and the subsequent analysis of Ainsworth and Brown (1933) provided justification for straightening and channelizing the river from Elephant Butte to Fort Quitman. The channelization was begun about 1933 and essentially completed in 1938. This reach of river is now artificially maintained as a water delivery and drainage canal.

Studying the remaining unchannelized reach between Fort Quitman and Presidio, Everitt (1993) concluded that the physical changes in the channel repre-

sented a complex chain of responses driven by deposition of excess sediment which the depleted river was no longer able to transport. He proposed a three-stage model for channel evolution. This model provides a conceptual basis for evaluating the relationship among the interdependent variables of declining stream flow, decreased flood magnitude and duration, floodplain aggradation, and channel capacity.

Everitt (1993) termed changes in channel width, channel depth, and channel cross-section area as “first-order responses” which began immediately after the flow regime changed, as excess sediment was deposited within the abandoned, oversized channel. Once channel capacity was reduced, over-bank flow resumed. These “second-order responses” included meander cutting, changes in the relationship between the main channel and its tributaries, and readjustment of channel gradient.

Deposition of sediment within the pre-dam channel of the Rio Grande occurred between 1915 and 1925 in the upstream end of the El Paso/Juarez valley (Ainsworth and Brown, 1933). The channel shrank in cross-sectional area, and overbank flooding did not occur during this time. Downstream from Fort Quitman, a similar pattern of infilling without overbank flooding occurred between 1915 and 1932. Photographs of the river taken during the U.S. Geological Survey hydrographic survey of 1901 depict a broad, shallow, sand-bedded channel downstream from Fort Quitman. Maps of the pre-dam river show a channel about 100 m wide. Aerial photographs taken in 1928 show that the channel had narrowed to about 30 m. Today, some of this old channel survives as oxbows that are lined with very old cottonwoods.

Beginning in 1925 near El Paso/Juarez and beginning in 1932 downstream from Fort Quitman, the channel’s flood capacity had sufficiently decreased such that the lower magnitude floods of this period again began to overtop its banks, depositing fine sediment across the valley floor (Figure 7). Floodplain inundation is a necessary process to cause meander cutoffs, and cutoffs became a renewed geomorphic process that had not occurred since 1915. Cutoffs occurred in each year in which flood discharges were large in relation to the shrunken channel.

The process of channel shrinkage was reversed in 1941 and 1942 when there were unusually large releases

from Elephant Butte reservoir. Losses were not great during these floods and the peak discharge at Presidio, upstream from the Río Conchos, was $145 \text{ m}^3/\text{s}$, which would have been about a 3-year recurrence flood prior to 1915. The high flows reestablished a larger channel cross-section that was narrower and deeper than the pre-dam channel (Figure 7). These changes occurred by erosion of the channel bed and by deposition of new floodplain sediments.

Channel infilling resumed after 1943 and continued downstream from Fort Quitman until 1963 (Figure 7). After 1963, sufficient aggradation had occurred that floodplain inundation and meander cutoffs again began to occur. In 1970, the Rio Grande channel was between 10 and 15 m wide (Everitt, 1993). The channel of the Rio Grande between Fort Quitman and Presidio is now about 90% smaller than the channel that existed in 1900.

Thus, the Rio Grande channel decreased in size by aggradation of the stream bed and deposition of bars inset within the former active channel and by floodplain

deposition. Everitt (1993) concluded that the post-1970 channel of the Rio Grande was approaching a balance between discharge and channel capacity. Thus, with the resumption of over bank flow the valley floor had resumed its function of storage of floodwater and sediment. Thus, deposition of sediment in the Rio Grande valley since about 1970 has occurred by concurrent deposition in the channel and on the floodplain such that the relationship between the two geomorphic features remains the same, and the geomorphic functionality of the river is relatively unchanged.

The contrast between the channelized reach between El Paso and Fort Quitman, and the natural reach from Fort Quitman and Candelaria, illustrates the consequences of different cultural treatments. Both reaches were initially similar in physical geometry and hydrology, and both experienced similar changes in flow regime following construction of Elephant Butte Dam. The channelized reach resembles a drainage ditch, dewatered much of the year, separated from its flood plain by steep banks, and with floodplain vegetation artificially main-

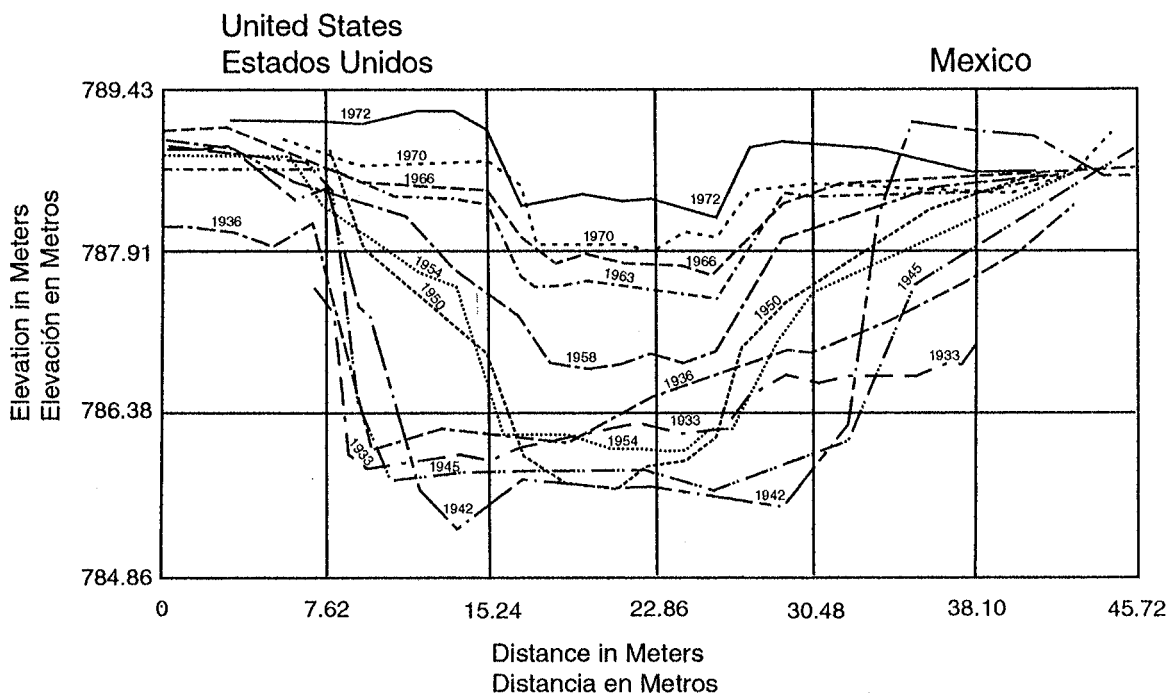


Figure 6. Graph showing composite cross sections of the Rio Grande channel at the gauging station above Rio Conchos, near Presidio, 1933-1974 (from Everitt, 1993, fig. 3).

tained by mowing. Beyond the levees, what remains of the cottonwood gallery forest is cut off from the river, its seeds falling on barren ground. Former tributaries have been cut off by erosion-control dams, isolating the river from its watershed. The riverine landscape has lost the physical continuity that once provided migration routes for riparian plants and animals, and the dynamic nature that once provided cycling and storage of water and nutrients.

Downstream from Fort Quitman, although severely depleted in stream flow, the river continues to be a functioning part of the landscape. Channel dimensions in some reaches have adjusted to the altered discharge so that a smaller river flows in a smaller channel in a relatively broader flood plain. As in pre-dam times, the river continues to meander in some places forming oxbows and in others braiding and forming islands, maintaining the topographic irregularities that provide habitat diversity. The mosaic of landscape elements necessary for

the foundation of a healthy riparian ecosystem is still present. Brushy banks with fallen trunks provide shaded scour holes for fish. Natural levees pond flood water beyond the channel, allowing it to percolate slowly back to the river, maintaining the shallow alluvial ground-water system and prolonging base flow. Broad overflow lands spread and filter water during high stages and flush accumulated salt from the soil. Here the river landscape retains both its longitudinal and lateral continuity, although there have been profound changes in the vegetation.

COCHITI TO ELEPHANT BUTTE

Changes in the Rio Grande between Cochiti and Bernalillo are similar to the pattern of bed degradation and narrowing that occurred soon after completion of Elephant Butte Dam. Richard's (2001) study of adjustments of the Rio Grande between Cochiti and Bernalillo demonstrated that continued lateral adjustments, includ-

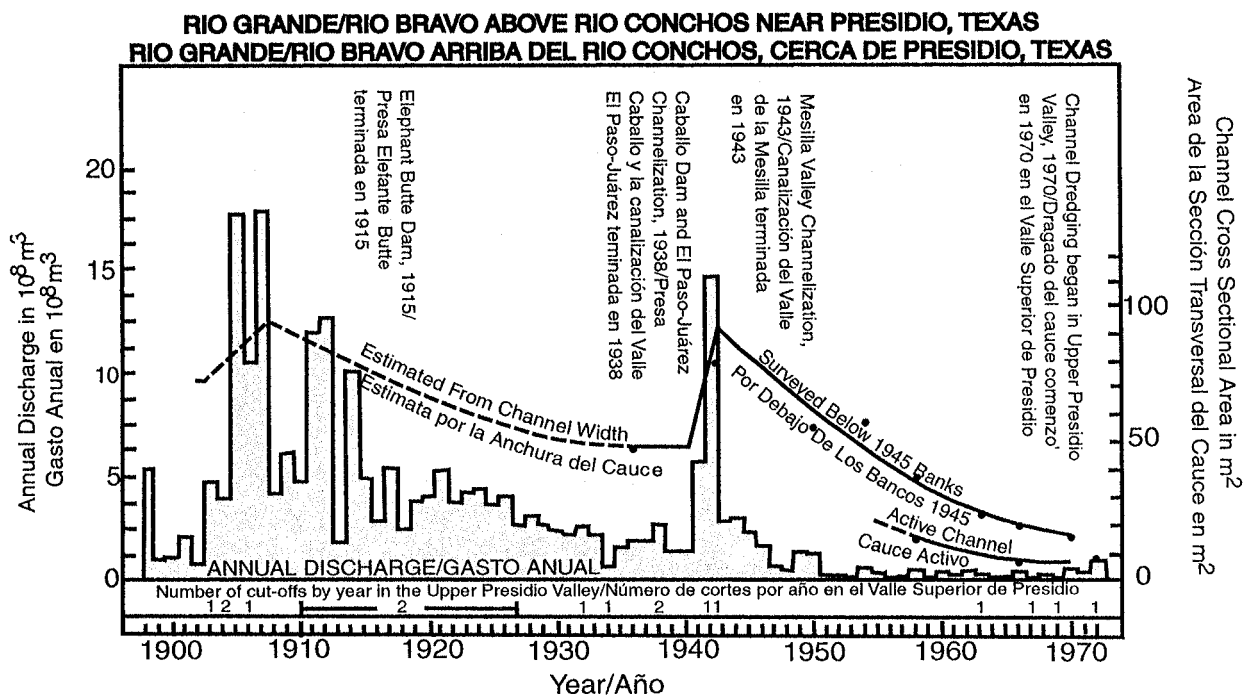


Figure 7. Graph showing time series of change in annual discharge and cross section at the gauging station above Rio Conchos, near Presidio, 1933-1974 (from Everitt, 1993, fig. 2).

ing narrowing and decreased lateral migration rates, occurred between 1918 and 1992. More rapid vertical adjustments occurred following construction of the dam in 1973. Prior to dam construction, the bed of the channel was primarily sand. The sandy bed of the channel responded to the temporal variability in sediment inputs by alternating aggradation and degradation; the net sum was gradual aggradation of the bed. Bed degradation began after Cochiti Dam was completed and up to 1.9 m of bed erosion occurred between 1972 and 1998. Following dam construction, the bed material between Cochiti and Bernalillo coarsened to gravel and cobbles (Richard, 2001).

Measurements from historic maps and aerial photographs indicate that as the peak discharges began to decrease (ca. 1930s) due to natural and anthropogenic factors the channel responded by narrowing, simplifying and reducing its rate of lateral migration (Figure 8). The width of the Rio Grande between Cochiti and Bernalillo decreased 60% prior to dam construction and by 1992 the channel was 70% narrower than in 1918. Also between 1918 and 1992 the channel planform shifted from a multi-thread braided configuration toward a single-

thread pattern as the number and size of mid-channel bars and islands decreased (Figure 9). Following dam construction, a meandering pattern became more pronounced as the sinuosity of the channel downstream from Cochiti increased slightly (Richard, 2001).

Richard (2001) concluded that attempts to “stabilize” the Cochiti reach of the Rio Grande through flood control, sediment detention, channelization, and bank stabilization succeeded in reducing the dynamism in both the inputs to the reach and in the responding form of the channel. Average lateral movements of the channel decreased from 27 m/year in 1918 to 5 m/year in 1992, and the active channel width has remained less than 100 m since 1985. Incision of the channel bed following construction of the dam disconnected the channel from the floodplain. The resulting narrow and deep configuration of the channel and reduced peak flows creates a situation in which even the highest flows no longer achieve bankfull conditions (Richard, 2001).

DISCUSSION

IMPLICATIONS OF HYDROLOGIC AND GEOMORPHIC HISTORY FOR RIVER RESTORATION OR REHABILITATION

We have shown that the physical attributes of the Rio Grande—its hydrology, sediment load, channel dimensions, and temporal variability of channel location—are much different than they were a century ago. Some segments are still evolving in response to past alteration in flood regime, depletion of flow, and deposition of sediment in reservoirs.

The aquatic and riparian ecosystems are also much different than they were a century ago. Ecosystem change is driven by the following variables:

- 1) Change in climate that affects the runoff and sediment flux.
- 2) Change in hydrologic and sediment regime caused by human activities.
- 3) Changes in the physical structure of the channel and floodplain. These changes are caused by changes in the flux of water and sediment and by direct manipulation of the channel or floodplain. The fundamental at-

tributes of physical structure are the cross-sectional form of the channel, the characteristics of the bed material and how it is organized, channel planform, channel gradient, and the relationship between the channel and its alluvial valley. These changes not only affect the distribution of aquatic habitats and the exchange rate of sediment between channel and alluvial valley but also the characteristics of nutrient spiraling.

- 4) Introduction of exotic species. In addition to many naturalizing herbs and grasses, there are 3 woody exotics that are expanding their range at the expense of native vegetation along the Rio Grande: Russian olive (*Eleagnus angustifolia* L.) in the upstream part of the northern branch, salt cedar in the southern part of the northern branch and the upstream part of the lower Rio Grande, and giant reed (*Arundo donax*) downstream from Presidio.

- 5) The internally-driven dynamics of ecosystems that cause some species to replace others over time.

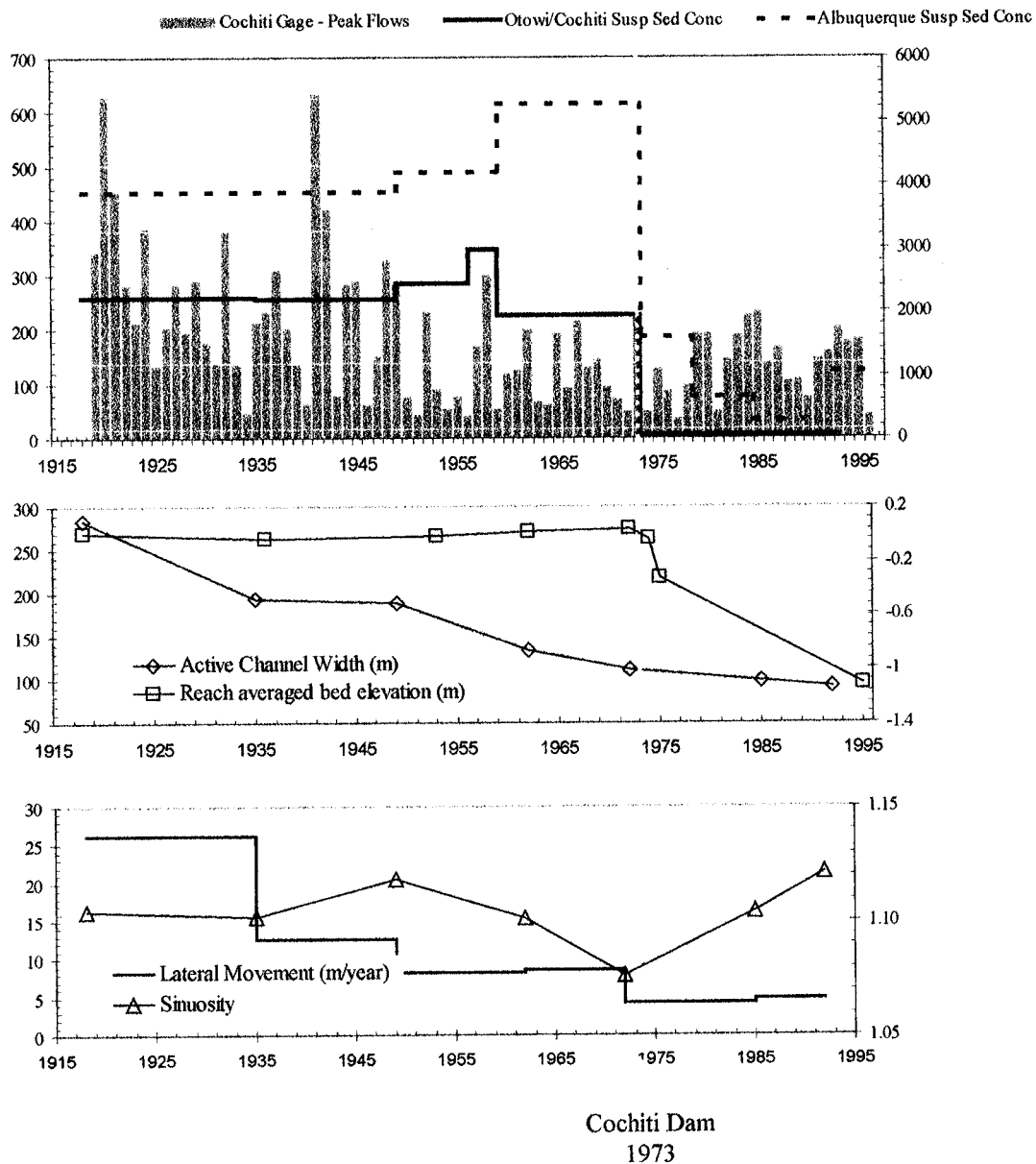


Figure 8. Summary of changes in the Rio Grande between Cochiti Dam and Bernalillo from 1918 to 1992.

Of these input variables, climate change is beyond the control of Rio Grande managers. Eradication of non-native species invasions and vegetation manipulation that alters the trajectory of ecosystem change are extremely difficult, although large-scale eradication of salt cedar has been conducted in parts of the Pecos River alluvial valley.

It is only by manipulating the runoff regime, sediment regime, or physically altering the channel or flood-

plain that we can alter the balance among competing species in a functioning ecosystem. We know we can do this, because we have already performed experiments on the Rio Grande. A hundred years of data on the 6 reaches of Table 5 provide case studies of how local riparian communities respond to different kinds of treatment under different local circumstances. Elsewhere, stream flow and sediment fluxes are being altered by dam reoperations in order to alter down stream ecosystems.

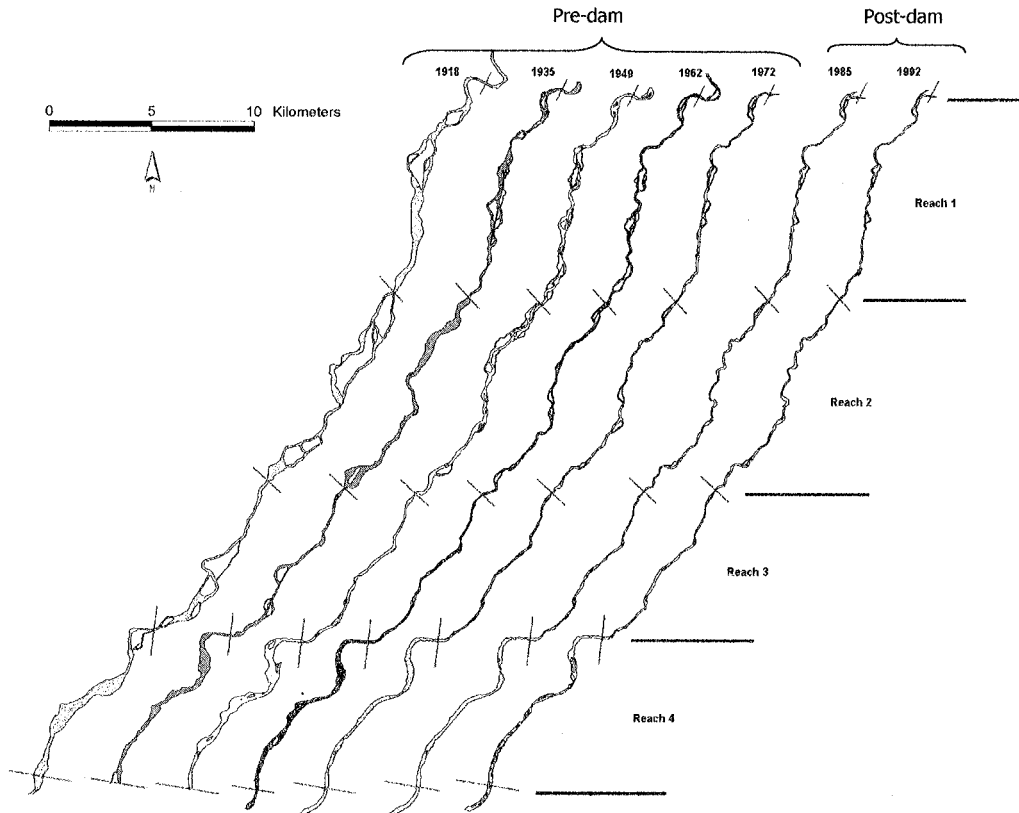


Figure 9. Planform maps of the active channel of the Cochiti reach for 1918 through 1992.

OTHER GOALS FOR ENVIRONMENTAL MANAGEMENT

Environmental management of the Rio Grande must be grounded in establishment of a set of well-defined goals for the future trajectory of ecosystem change on each segment. Ecosystem restoration is defined as “the return of an ecosystem to a close approximation of its condition prior to disturbance” (National Research Council, 1992). There also are other possible goals for aquatic ecosystems, including reclamation, rehabilitation, mitigation, and creation (National Research Council, 1992). *Reclamation* is the process of adapting a wild or natural resource to serve a utilitarian human purpose. Thus, this term is reserved for activities such as converting native floodplain ecosystems to agricultural uses. *Rehabilitation* is a term used primarily to indicate putting a natural resource back into good condition or working order. *Mitigation* is typically defined as alleviating any or all detrimental effects arising from a specific human

activity. *Creation* is the bringing into being of a new ecosystem that previously did not exist at the site. Environmental management goals might include any of those listed above. Choice of goals, on a segment by segment basis, is an effort in policy development, will inevitably be based on dialogue among river stakeholders, and is necessarily political.

Disturbances to the hydrologic regime of the Rio Grande began hundreds of years ago and are significant. Restoration of the Rio Grande’s northern branch to its condition in 1900 would require dam decommissioning and the abandonment of most irrigated agriculture in southern Colorado, New Mexico, the El Paso/Juarez valley, and the Presidio valley. Restoration would also require removal of levees, rehabilitation of channelized sections, and relocation of large numbers of people from

the historic floodplain of the rivers, especially in El Paso and Ciudad Juarez. Political consensus to undertake such a comprehensive program of river restoration probably does not exist in either the United States or Mexico. Thus, it is essentially impossible to restore most of the river.

Goals for the Rio Grande might include (1) rehabilitation to some post-1915 condition, although the channel was not in equilibrium with its floodplain for most of this time, (2) rehabilitation so that the channel and alluvial valley have a broader suite of ecological processes and attributes similar to the pre-disturbance river, (3) mitigation by maintenance of a new ecosystem, with or without salt cedar, that is adjusted to a specified range of flood flows and annual flows, (4) mitigation to the level of ecosystem function necessary to recover endangered species, or (5) acceptance of the riverine ecosystem as it is today. The identification of the appropriate goal depends on a precise identification of the natural and human values that would be improved and degraded if the present ecosystem were changed.

There is probably no single environmental management goal that is appropriate for all of the Rio Grande. Each goal described above is associated with its own economic cost, and achievement of political consensus on any environmental management goal is difficult to achieve. Knowledge of the magnitude of twentieth cen-

tury environmental change does not necessarily mean the trajectory towards restoration will follow the same path and the trajectory of historical change. Where channels have significantly narrowed, become disconnected from their floodplains, and overgrown with salt cedar, the question remains whether reintroducing more natural water and sediment fluxes will immediately reverse undesired historical changes. Restoration science is not yet able to predict these trajectories of system recovery.

In the face of such uncertainty, pursuing uniform basin-wide rehabilitation goals is essentially impossible. Is it better to ask where in the basin can undesired historical changes be efficiently reversed? Where in the basin will the native riverine ecosystem respond most favorably to reversal of historical changes in the physical environment? Where can the historical changes of water and sediment flux be feasibly reversed and at what political cost? Where is the greatest need for rehabilitation? Answers to each of these questions can only be provided by considering what is feasible and possible in each segment of the river. Only then can one examine how much water is available for redistribution to environmental objectives and develop an allocation system that recognizes the needs of the natural riverine ecosystem and the physical template within which it has developed.

CONCLUSION

The extent of changes to the water and sediment flux are so great, and the extent of changes to the physical system of the channel and floodplain are so extensive, that comprehensive restoration of the Rio Grande is impossible. Priorities must be established wherein different environmental management goals are established for different segments of the river system. Establish-

ment of these priorities is inevitably a political process, wherein the role of the scientific community is to present a clear picture of the magnitude of transformation of the present riverine ecosystem from its pre-disturbance condition and the activities necessary to rehabilitate the ecosystem to varying degrees or to reverse undesired changes in the physical or ecological system.

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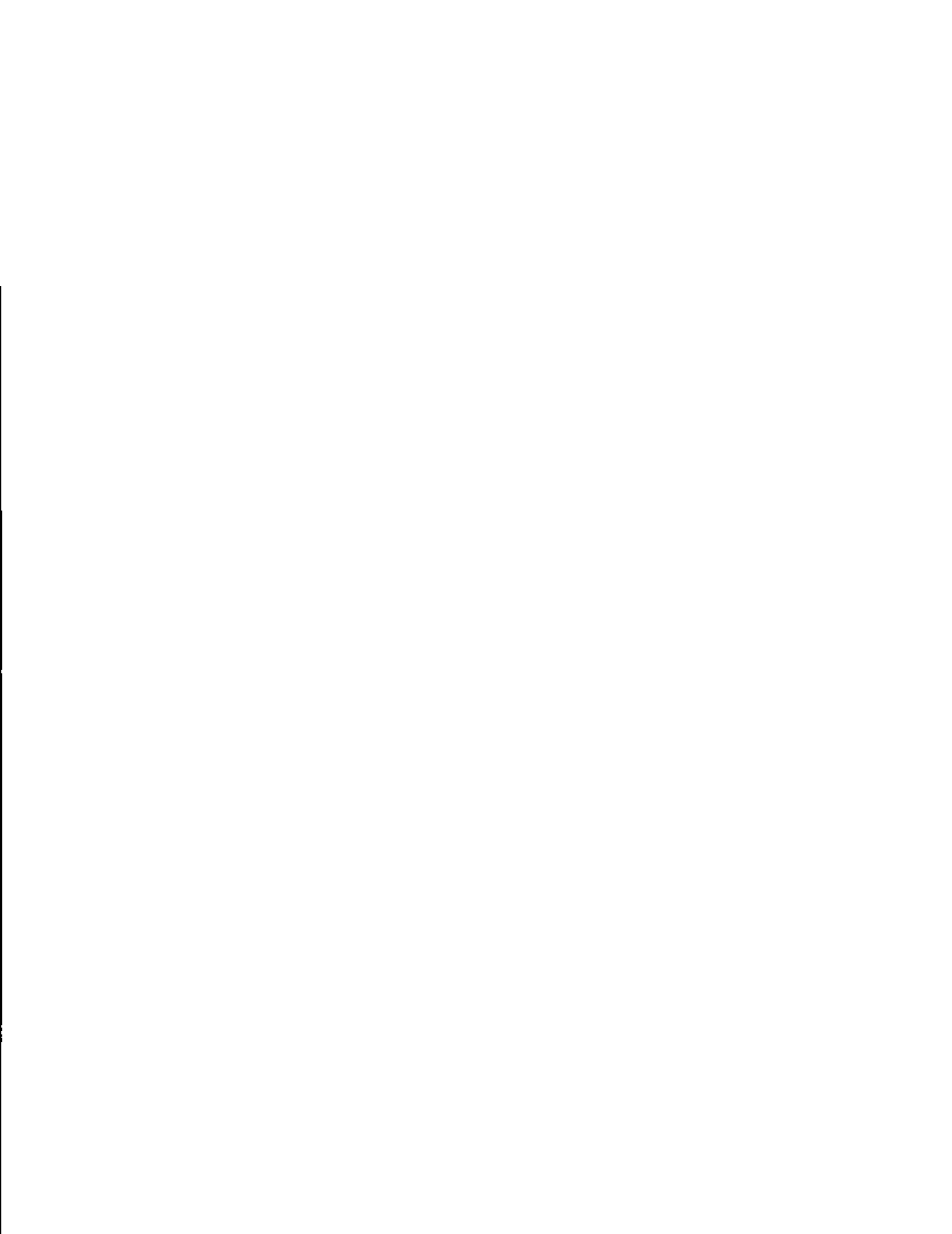
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GAMMARID AMPHIPODS OF NORTHERN CHIHUAHUAN DESERT SPRING SYSTEMS: AN IMPERILED FAUNA

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Mark E. Gordon and Gordon Warrick

ABSTRACT

Gammarid amphipods of the *Gammarus pecos* complex Cole, 1985 are restricted to euryhaline desert spring systems in the Pecos River Valley of New Mexico and Texas. Within the past 35 years, complete loss and diminution of spring flows, exacerbated by regional drought conditions and local groundwater withdrawals, are implicated in the extirpation of 2 isolated populations of *G. desperatus* in New Mexico, and the dramatic decline of the *Gammarus* sp. form "C" from the Phantom Lake Spring system in Texas. The distribution and abundance of gammarid amphipods

was determined from benthic samples collected in May 2001 from 7 sites in the study area. Gammarid densities were highest in shallow, low velocity habitats with aqueous silts. Three new gammarid populations were documented from 2 springs in Reeves County, Texas, and 1 site in Eddy County, New Mexico. This study illustrates the need for comparative morphological and biochemical genetic studies of the *G. pecos* complex to clarify outstanding taxonomic relationships within this group.

INTRODUCTION

Members of the *Gammarus pecos* complex Cole, 1985 represent endemic species geographically isolated in euryhaline desert spring systems of the Pecos River Valley of New Mexico and Texas. These freshwater amphipods are likely derived from a widespread progenitor marine amphipod that was isolated inland during recession of the Late Cretaceous epicontinental sea (ca. 66 mya) (Bousfield, 1958; Holsinger, 1976). Speciation within this complex likely occurred as a result of local adaptive variation in response to ecological constraints imposed by diverse aquatic environments on amphipod populations further isolated during progressive climatological changes that ensued in the late Pleistocene to early Holocene. Such models of island and vicariant biogeography are proposed for a diversity of invertebrate taxa within arid ecosystems of southwestern North America (e.g., Peracarida crustaceans [freshwater isopods, see Bowman, 1981; hyalellid amphipods, see Thomas et al., 1994]; prosobranch snails, see Hershler et al., 1999; pulmonate land snails, see Bequaert and Miller, 1973, Metcalf and Smartt, 1997).

Based on percent similarities in Mann-Whitney U tests for 20 morphological traits from 7 populations of gammarid amphipods, Cole (1985) identified the *G. pecos* complex as a group of morphologically similar species that are endemic to Chihuahuan Desert spring systems of the Pecos River Valley in southeastern New Mexico and western Texas. Morphological character combinations unique to this group include: non-calceolate antennae with spine(s) on the first peduncular article of the antennule; mandibular palps bearing C-setae; setiferous coxal plates I-IV; and narrow oostegites (brood plates). Currently this complex is understood to consist of 3 described species (Cole and Bousfield, 1970; Cole, 1976, 1981), 2 populations representing undescribed species, and 2 morphotypes of undetermined taxonomic affinity (see Cole, 1985).

Cole (1985) emphasized numerous unresolved taxonomic relationships within this complex based on marked intra- and inter-population variation of morphological characters and body sizes among West Texas gammarids.

Formulation of effective conservation measures or intensive ecological assessments would be premature until outstanding taxonomic affinities of the *G. pecos* complex have been clarified. Morphological assessment of the *G. pecos* complex, considered in combination with ongoing genetic studies, can fill a critical gap in our current knowledge of these endemic taxa: "Which species or significant population segments merit conservation management?" Resource agencies charged with protection and stewardship of these amphipods have no baseline data from which to assess conservation status and threats, or to prescribe management options for this group.

Amphipods are vital links in aquatic food webs and play critical roles in nutrient processing of aquatic ecosystems (Gee, 1988; Pennak, 1989). Due to their acute sensitivity to aquatic conditions (Covich and Thorpe, 1991), gammarid amphipods can be considered ecological indicators of ecosystem health (Lackey, 1995) and integrity (Callicott, 1994). The current rate of imperilment of amphipods of the *G. pecos* complex is alarming and provides testimony to the deterioration of aquatic conditions of the Pecos River Basin of New Mexico and Texas.

No gammarids were observed during preliminary status surveys of Phantom Lake Spring and its canal system in March and June 2000, which suggested that *Gammarus hyalelloides* Cole, 1976 and the undescribed

Gammarus sp. form "C" were possibly extirpated from this aquatic system. Although the location of Cole's (1985) *Gammarus* sp. form "M" population could not be reconciled from locality descriptors while in the field, the ample evidence of spring head drying and defunct irrigation canal systems observed in the general vicinity of Phantom Lake Spring suggested that this gammarid morphotype may be extirpated as well. These preliminary findings prompted this study.

Specific causes for the suspected extirpation of *Gammarus* sp. form "C" from the aquatic environs of Phantom Lake Spring appeared directly related to reduced spring discharge. Diminution of Phantom Lake Spring discharge has been attributed to depletion of the Toyah Basin Aquifer from groundwater withdrawals under ongoing regional drought conditions (Ashworth et al., 1997; Schuster, 1997; Sharp et al., 1999; Allan, 2000). Similar factors were largely responsible for the extirpation of 2 isolated populations of *Gammarus desperatus* Cole, 1981 in New Mexico (Cole, 1981, 1985).

We present preliminary findings from a macroinvertebrate survey to document the current status of gammarid amphipods from desert spring systems in the Pecos River Valley of New Mexico and Texas. This study illustrates the need for comparative morphological and biochemical genetic studies of the *G. pecos* complex.

STUDY AREA AND METHODS

The distribution and abundance of gammarid amphipods was determined from benthic samples collected in May 2001 from 7 sites in the study area of southeastern New Mexico and West Texas (Figure 1). Amphipod densities were determined from 3 random benthic collections per site using a stainless steel mesh sampler (sample area = 80 cm²). Benthos were preserved in 95% ethanol for enumeration by taxa in the lab. Habitat parameters measured at each sample site included water depth (± 1 cm; metric topset rod), velocity (cm/sec; Marsh-McBirney Flowmate®), and substrata following the general scheme of Cummins' (1962) index of dominant grain size as clay, silt, sand, gravel, and mixed. Vegetation, including submerged macrophyte beds and detrital plant debris, was included

as a "substrate" category since amphipods of this complex typically frequent this cover type. Physicochemical data (i.e., water temperature [°C], pH, dissolved oxygen [DO; mg/l, % saturation], salinity [ppt], specific conductance [μ S/cm] and total dissolved solids [TDS; ppt]) presented herein were compiled from published accounts and agency reports due to equipment failure in the field.

Live *Gammarus* for ongoing morphological and biochemical genetic studies were collected with a dip net by sweeping submerged macrophyte beds, by sieving aqueous silt, or from the underside of coarse substrata by washing into a tray. To quantify genetic and morphological variation within a gammarid population,

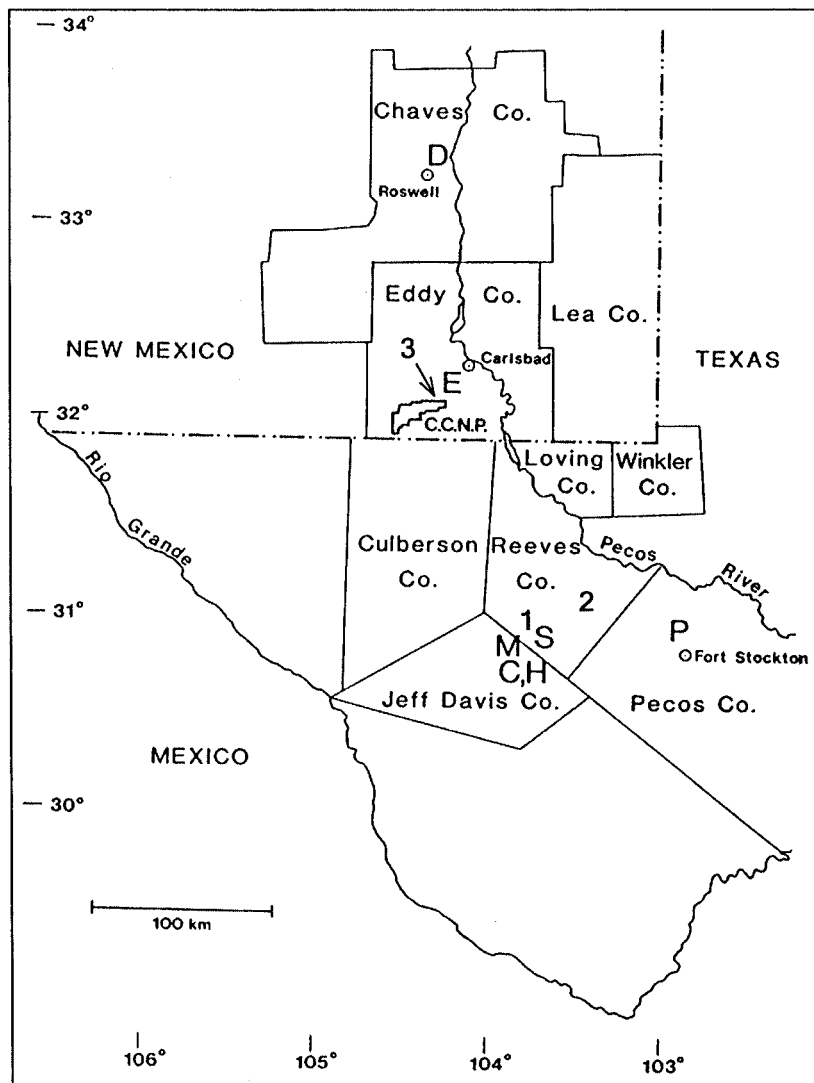


Figure 1. Historic and current distribution of gammarid amphipods of the *Gammarus pecos* complex Cole, 1985 in the Pecos River Valley of New Mexico and Texas. Population designations from modified map per Cole (1985.): "D", *G. desperatus*, Roswell, Chaves Co., NM (extant at Bitter Lake National Wildlife Refuge, extirpated from Lander Springbrook and North Spring); "E", *Gammarus* sp., "in Carlsbad" or "in Carlsbad Caverns National Park", Eddy Co., NM; "P", *Gammarus pecos*, Diamond Y Spring, Pecos Co., TX; "S", *Gammarus* sp., San Solomon Spring, Toyahvale, Reeves Co., TX; "H", *Gammarus hyalelloides*, Phantom Lake Spring, Jeff Davis Co., TX; "C", *Gammarus* sp., Phantom Lake Spring canal system, Jeff Davis Co., TX; "M", *Gammarus* sp., "3.5 miles west of Toyahvale" or "350 m north of Phantom Lake Spring", Jeff Davis Co., TX. Numeric population designations: 1, *Gammarus* sp., Giffin Spring, Toyahvale, Reeves Co., TX; 2, *Gammarus* sp., East Sandia Spring, Reeves Co., TX (0.5 mi. east of Brogado, TX, and 0.3 mi. south of state route 17); 3, *Gammarus* sp., Sitting Bull Spring, Eddy Co., NM.

representative voucher material was collected from diverse habitat types employing subsampling in large spring systems (e.g., Diamond Y Preserve, San Solomon Spring) or from populations with geographically isolated populations (i.e., Bitter Lake National Wildlife Refuge). This sampling protocol yielded adequate voucher material for intra- and interspecific

genetic and morphological studies from multiple subsamples for 7 gammarid populations. At least 50 amphipods from each collecting locality were frozen in liquid nitrogen immediately upon collection for genetic study. Specimens retained for morphological analysis (N = 50 amphipods/locality) were preserved in 95% ethanol.

RESULTS

Our survey documented 3 previously undetected gammarid populations from Giffin and East Sandia springs, Texas, and Sitting Bull Spring, New Mexico, that morphologically are referable to the *G. pecos* complex (Cole, 1981, 1985). It remains unclear to us whether the Sitting Bull Spring gammarid population is referable to Cole's (1985) *Gammarus* sp. form "E", or may actually represent a previously undocumented population, as Cole listed two possible localities for the latter taxon: "A third unnamed population occurs [or once occurred] in Carlsbad Cavern National Park, or the town of Carlsbad..." Notwithstanding, preliminary analysis indicates that the Sitting Bull Spring population appears morphologically distinct from the nearest gammarid populations in New Mexico (*G. desperatus*) and Texas (*G. pecos*). Morphological studies are ongoing to describe the Sitting Bull Spring gammarid, and to compare within- and among-population variation of morphological characters in this species complex.

No gammarids were observed in Phantom Lake Spring or in the downstream canal system during the May 2001 inventory. All lateral canals in the immediate area were either dry or dysfunctional; this further confirmed our March and June 2000 observations that *Gammarus hyalelloides* and the morphotype *Gammarus* "C" were in all likelihood extirpated from the Phantom Lake Spring system. However, sampling during a visit in November 2001 by Lang and Berg revealed *G. hyalelloides* from the cave mouth pool, where intensive surveys in March and June 2000, and May 2001, failed to yield even the slightest evidence of an extant gammarid population. We also found live *G. hyalelloides* in hypogean habitats at 3-4 meters inside the cave's entrance during the November 2001 survey.

Table 1 presents amphipod densities (\pm SE) and flow conditions (depth and velocity; \pm SE) measured at sample sites. Physicochemical parameters compiled from published records and agency reports (Table 2) show highly variable chemical environments with ionic concentrations of these fresh to moderately saline desert spring systems determined largely by the underlying karst stratigraphy.

Amphipod densities were highest in the Diamond Y Draw spring system (Euphrasia Spring, *Gammarus pecos*, = 8,042 amphipods/m²) where mean water depths and velocities measured at 3 sites ranged from 0.06-0.16 m and 0.03-0.06 m/sec., respectively. The gammarid population in the upper rheocrene of Sitting Bull Spring was the least dense with 125 amphipods/m². Low flow conditions in the main canal of San Solomon Spring (velocity = 0; depth = 0.09 m \pm 0.02 SE) likely account for the high mean density of *Gammarus* sp. form "S" (6,833 amphipods/m²) during seasonal draw down for swimming pool maintenance at Balmorhea State Park. Mean density of *G. desperatus* (575 amphipods/m²) from Sago Spring is considered lower than expected since this estimate was derived from artificial tile samples (Lang, unpublished data).

Hyaellid amphipods are referred to herein as *Hyaella* sp. since ongoing taxonomic research continues to identify genetically and morphologically distinct populations once considered as a single ubiquitous species, *Hyaella (Hyaella) azteca*, in the southwest United States (see references in Baldinger et al., 2000; Duan et al., 2000). *Hyaella* occurred syntopically with *Gammarus* sp. at East Sandia (1,083 *Hyaella* sp./m²) and Euphrasia (125 *Hyaella* sp./m²) springs (Table 1). Densities of *Hyaella* sp. were high-

Table 1. Mean density of hyalellid and gammarid amphipods (no./m²) and mean water depth (m) and mean velocity (m/sec.) measured at benthic sample sites (N = 36 grabs) in desert spring systems of the Pecos River valley of New Mexico and Texas, May 2001.

Taxon	Site	Amphipod Subsample	Amphipod Density ($\bar{X} \pm SE$)	Water Depth ($\bar{X} \pm SE$)	Water Velocity ($\bar{X} \pm SE$)
Texas					
<i>Gammarus hyalelloides</i>	Phantom Lake Spring	spring head pool	0	0.40 ± 0.04	no measurable flow
<i>Gammarus</i> sp. form "C"	Phantom Lake Spring	lateral canal	0	dry	-
<i>Gammarus</i> sp. form "S"	San Solomon Spring	swimming pool	0	0.18 ± 0.04	no measurable flow
<i>Gammarus</i> sp. form "S"	San Solomon Spring	main canal	6833 ± 5416	0.09 ± 0.02	no measurable flow
<i>Gammarus</i> sp. form "S"	San Solomon Spring	ciénega outflow	-	-	-
<i>Gammarus</i> sp.	Giffin Spring	spring head pool	0	0.26 ± 0.06	no measurable flow
<i>Gammarus</i> sp.	Giffin Spring	rheocrene	1167 ± 730	0.19 ± 0.09	0.12 ± 0.04
<i>Gammarus</i> sp.	East Sandia Spring	rheocrene	4625 ± 804	0.10 ± 0.03	0.14 ± 0.08
<i>Hyalella</i> sp.	" "	" "	1083 ± 1083	" "	" "
<i>Gammarus pecos</i>	Diamond Y Draw	rheocrene near source	2208 ± 1585	0.16 ± 0.04	0.03 ± 0.02
<i>Gammarus pecos</i>	Diamond Y Draw	Euphrasia Spring	8042 ± 7229	0.06 ± 0.00	0.06 ± 0.03
<i>Hyalella</i> sp.	" "	" "	125 ± 72	" "	" "
<i>Gammarus pecos</i>	Diamond Y Draw	John's Pool	2708 ± 1381	0.09 ± 0.02	0.04 ± 0.01
New Mexico					
<i>Gammarus</i> sp.	Sitting Bull Spring	lower rheocrene	4542 ± 2361	0.09 ± 0.01	0.09 ± 0.02
<i>Gammarus</i> sp.	Sitting Bull Spring	upper rheocrene	125 ± 72	0.12 ± 0.03	0.02 ± 0.01
<i>Hyalella</i> sp.	" "	" "	3167 ± 1601	" "	" "
<i>Gammarus desperatus</i> *	Bitter Creek	Lost River pool	4250 ± 683	0.16 ± 0.04	0.12 ± 0.03
<i>Gammarus desperatus</i> *	Bitter Creek	Sago Spring head	575	0.18 ± 0.04	0.08 ± 0.06
<i>Gammarus desperatus</i> *	Bitter Creek	Sago Spring run	3750 ± 735	0.07 ± 0.06	0.18 ± 0.05
<i>Gammarus desperatus</i>	Bitter Creek	Unit 6	167 ± 110	0.11 ± 0.02	0.16 ± 0.04

*Density estimates derived from previous study (Lang, unpubl. data).

Table 2. Physicochemical data for desert spring systems in the Permian Basin where gammarid amphipods of the *Gammarus pecos* complex Cole, 1985 occur in New Mexico and Texas.

Site	Temperature °C	pH units	Specific Conductance $\mu S/cm$	Salinity ppt	TDS mg/l	DO % saturation	DO mg/l
Lander Springbrook, Chaves Co., NM ^{1,2}	18–22	7.1–7.2	—	4.4–5.7	—	—	—
North Spring, Chaves Co., NM ²	19.0–20.5	7.2	17,600	5.7	—	—	—
BLNWR, Bitter Creek, Chaves Co., NM	7.6–25.2	6.9–8.7	8044–10,538	4.9–5.3	5.5–6.7	8.0–191.1	0.8–15.9
BLNWR, Sago Spring, Chaves Co., NM	16.0–21.5	7.0–7.3	6500–8100	4.3–5.1	5.1–5.8	10.0–185.0	2.5–14.8
BLNWR, Unit 6, Chaves Co., NM	22.8	7.3	5367	2.9	3.5	70.2	7.4
Sitting Bull Spring, Eddy Co., NM	14.0–18.5	—	—	—	—	—	—
Diamond Y Draw (Sta. 2), Pecos Co., TX ³	20.6	7.1	—	3.6	—	—	6.7
East Sandia Spring, Reeves Co., TX ³	19.8	7.2	—	2.2	—	—	8.2
Giffin Spring, Reeves Co., TX	15.5–17.5	—	—	—	—	—	—
San Solomon Spring, Reeves Co., TX	25.0	7.2	—	1.8	—	—	1.9
PLS, Jeff Davis Co., TX ³	24.7	6.9	—	1.9	—	—	1.9
PLS, below weir (Sta. 1), Jeff Davis Co., TX ³	19.1	6.5	—	4.2	—	—	8.8

¹ Noel (1954); ² Cole (1981); ³ Hubbs (2001)

BLNWR = Bitter Lake National Wildlife Refuge; PLS = Phantom Lake Spring

est (3,167 amphipods/m²) at the upstream sample site in Sitting Bull Spring. There was no evidence of geographic overlap between these amphipod genera at the downstream site in this system. We note similar distributional patterns of amphipods at Bitter Lake National Wildlife Refuge (BLNWR) where *G. desperatus* shows affinities for shallow lotic habitats near thermally stable spring sources, while *Hyaella* occurs in more lentic eurythermal waters and deep aqueous silts.

The abundance of amphipod taxa was variable across the spectrum of substrate types sampled (N = 36 benthic grabs) throughout the study area (Figure 2). Sampling inefficiency likely accounts for the high outlying densities of *Gammarus* spp. in clay (22,500

amphipods/m²) and gravel (18,280 amphipods/m²) substrata since sample effort in these indurate substrate types was limited to a single benthic grab. Small sample size (N = 3) may likewise explain the high density of *Gammarus* (3,667 amphipods/m²) found in sand. Non-indurate substrata afforded by loosely compacted, aqueous silts (N = 16 grabs) and vegetation (i.e., macrophyte beds, detritus; N = 7 grabs) were inhabited by both amphipod genera, but to a lesser degree than mixed substrates (N = 8 grabs) where *Hyaella* (1,078 amphipods/m²) and *Gammarus* (2,406 amphipods/m²) were abundant. Gammarid amphipods were particularly abundant beneath large stones in lotic habitats that were difficult to sample by our quantitative methods.

DISCUSSION

All desert spring systems surveyed in this study occur within an area once overlain by Permian seas (Hills, 1942) where the underlying geology of these karst lands consists of dissolute evaporite rock (White et al., 1995; Martinez et al., 1998). Several basins in

the lower Pecos River Valley are described in the study area, namely the Roswell Artesian Basin of New Mexico, the Toyah Basin of West Texas (Hill, 1900), and the Delaware Basin in both states (Lang, 1937). Gammarid populations of this complex occur in

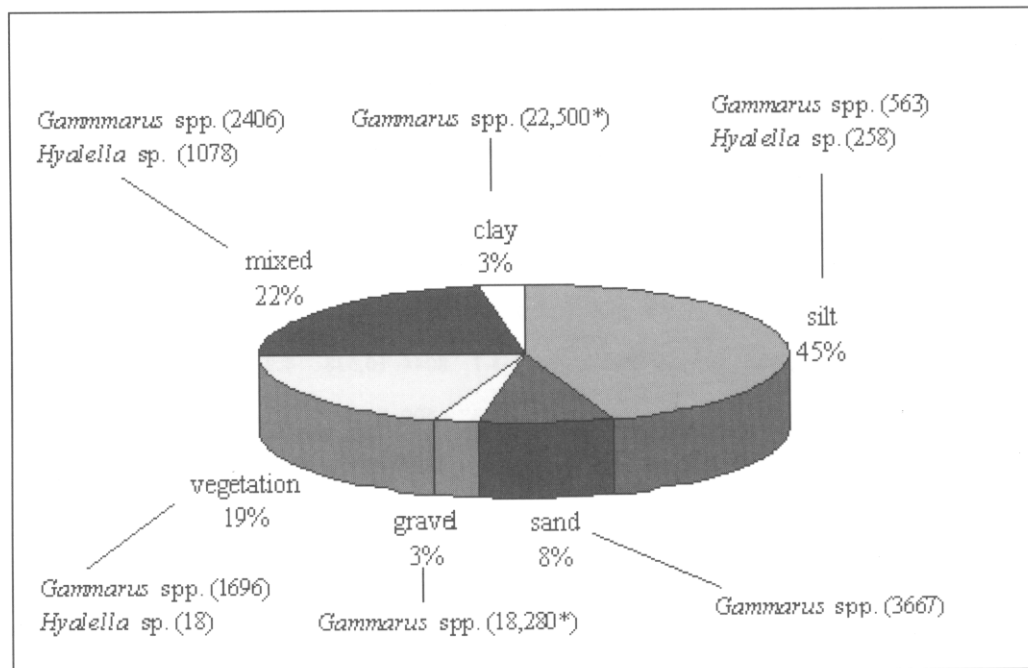


Figure 2. Mean density of Amphipoda (#/m²) by percentage substrate types sampled (n=36 benthic grabs) in desert spring systems surveyed in the Pecos River Valley of New Mexico and Texas, May 2001. (*Denotes sample bias from a single benthic grab.)

sulfatochloride, carbonate-rich spring systems of variable salinities and alkalinities over small spatial scales that Cole (1985) considered “fresh-to-miohaline” (Table 2). There are numerous reports that such spatial diversity of hydrochemical habitats and lithological features in karst basins of the eastern United States and Central Europe leads to habitat segregation, migration barriers, and geographical differences in genetic and morphological variation among gammarid amphipods in discrete, highly-structured habitats (Gooch and Hetrick, 1979; Foeckler and Schrimppff, 1985; Gooch, 1989, 1990; Sarbu et al., 1993). Similar physical and ecological constraints may be responsible for speciation within the *G. pecos* complex.

Within the past 35 years, diminution or complete loss of spring flow, exacerbated by regional drought conditions and local groundwater withdrawals, is implicated in the extirpation of 2 isolated populations of gammarid amphipods in New Mexico (*G. desperatus* from Lander Springbrook and North Spring [Cole, 1981, 1985]), and *Gammarus* sp. form “C” from the seasonally dewatered and impoverished aquatic conditions of the Phantom Lake Spring canal system in Texas (Allan, 2000). Gammarid amphipods in the Phantom Lake Spring canal were previously documented in 1995 (Winemiller and Anderson, 1997).

The enigmatic recurrence of *G. hyalelloides* at Phantom Lake Spring pool in November 2001, where repeated survey effort from March 2000 to May 2001 implied that the species was possibly extirpated, may be attributed to a number of factors. Preliminary survey effort focused on epigeal habitats at the spring head pool since access to hypogean waters was limited. However, recent scuba surveys did not document gammarid amphipods from hypogean habitats deep within the Phantom Lake Spring cave system where several aquatic troglobites (i.e., the isopods, *Lirceolus cocytus* [Lewis, 2001] and *Cirolanides texasensis*, and an undescribed cave-adapted amphipod) are reported (J. Krejca, University of Texas at Austin, pers. comm.). Aquatic surveys may have simply over-looked shallow lotic habitats near the cave’s entrance that interface between epigeal waters and deeper hypogean habitats. Similar cave streams are colonized by *Gammarus minimus* in highly structured karst areas of the eastern United States (Holsinger and Culver, 1970; Gooch and Hetrick, 1979). Secondly, it

is possible that under diminished spring flows and impoverished aquatic conditions of Phantom Lake Spring (Allan, 2000), that *G. hyalelloides* was either at very low densities or actually extirpated from epigeal habitats during the period March 2000 to May 2001. Efforts to sustain the endangered fish community in Phantom Lake Spring through installation of a pumping system commenced in June 2000 with construction of a sand-bag dam in the cave mouth to maintain adequate surface water elevation in the spring pool. The pump was activated in May 2001 and began cycling water from the cave to the spring pool. Such conservation practices may have serendipitously prevented the extirpation of *G. hyalelloides* at Phantom Lake Spring by replenishing the epigeal pool population with individuals of a hypogean source population of *G. hyalelloides* from inside the cave.

The March 2000 Sandhill Fire at BLNWR has resulted in demonstrable effects on abiotic conditions (e.g., depressed dissolved oxygen levels [0.7 mg/l], increased temperature, spikes in salinity and pH) of Bitter Creek, which harbors 1 of 3 known *G. desperatus* populations on BLNWR (Lang 2001). Preliminary findings implicate increased levels of post-fire polycyclic aromatic hydrocarbons as a possible adverse short-term effect on localized populations of *G. desperatus* in Bitter Creek. All described amphipods of this complex are considered Federal species of concern. *Gammarus desperatus* is the only species that has formal protection as a state Endangered species (NMGF Regulation 657); and has been recently proposed for Federal listing as endangered under the Endangered Species Act (67 Federal Register 6459-6479).

Although direct causes responsible for the extirpation of gammarid amphipods from North Spring and the seasonal disappearance from the Phantom Lake Spring system remain undetermined, altered hydrologic conditions of regional groundwater aquifers affecting these desert springs are implicated. Reduced flow regimes can significantly alter the physicochemical balance of a lotic ecosystem (Hynes, 1970), resulting in an altered hydrologic regime, disruption of complex ecological relationships, and adverse consequences for aquatic fauna of desert spring ecosystems (Bowles and Arsuffi, 1993; Mehlhop and Vaughn, 1994; Hubbs, 1995, 2000, 2001).

Local adaptive variation (genetic and morphological) of gammarid amphipods is reported from diverse aquatic ecosystems in response to local environmental conditions and regional geohydrology (Gooch and Hetrick, 1979; Gooch, 1989, 1990). In some instances morphological differentiation in hyalellid and gammarid amphipods has been attributed to genetic differentiation (Kane et al., 1992; Sarbu et al., 1993; Müller et al., 1999; Duan et al., 2000). Physiologically stressful conditions, frequently caused by anthropogenic activities, can select for certain genotypes in hyalellid and gammarid amphipods (Guttman, 1994; Duan et al., 1997; Hogg et al., 1998, 1999). Such selection can result in highly localized populations consisting of stenotropic demes acutely adapted to a narrow range of abiotic conditions (e.g., physicochemical gradients) and hydrologic regimes. Under extreme environmental stress isolated amphipod populations may lack sufficient genetic diversity to cope with stochastic fluctuations in the aquatic environment and are susceptible to extirpation events.

From a conservation perspective, genotypic variability provides populations with genetic plasticity to cope with environmental perturbations. Amphipod species with low genotypic variation and marked morphological differentiation are usually highly adapted to specific natural habitat conditions (e.g., *G. pecos* complex), but may be less capable of surviving long-term environmental change (Foeckler and Schrimpf, 1985; Duan et al., 1997; Hogg et al., 1998, 1999). Evolutionary consequences must be considered in the evaluation of natural or anthropogenic environmental stressors in order to direct conservation efforts that focus on not only a typological approach for species protection, but also consider important population segments as significant evolutionary units that likewise merit preservation (Rojas, 1992; Nee and May, 1997).

Cole (1985) documented marked overlap in body size and morphological characters between geographi-

cally proximate gammarid populations "M", "S" and "C" of West Texas. The numerous unresolved taxonomic affinities within the *G. pecos* complex highlight the need for phylogenetic assessment of this group focusing on comparative morphological and biochemical genetic studies. Recent genetic studies of hyalellid amphipods (Guttman, 1994; Duan et al., 1997; Thomas et al., 1994; Hogg et al., 1998; McPeck and Wellborn, 1998) indicated that lack of phylogenetic research has impeded studies of *Hyaella* comparative biology, biogeography, and evolution (Duan et al., 2000).

Morphological discrimination of amphipod species requires analysis of whole animal morphology, proportional metrics of diagnostic character combinations and structural meristics of sexually mature males and females to distinguish cryptic species (Holsinger, 1967, 1976; Müller et al., 1999). We are currently engaged in such studies, and limit our comparison of characters examined from new gammarid populations discovered during this survey (i.e., Giffin, East Sandia, and Sitting Bull springs) with traits diagnostic for the complex (see Cole, 1985). Biochemical genetic analyses using allozyme electrophoresis to compare within- and among-population variation are ongoing. While preliminary results of these genetic studies indicates significant within-population variation with 75% of the populations showing heterozygote deficiencies (Gervasio, unpubl. data), ongoing analyses will characterize the partitioning of genetic variation across spatial scales.

Once taxonomic boundaries of this complex are identified, resource agencies and private land stewards will have a solid baseline for evaluating broad-scale environmental trends and assessment of threats due to environmental degradation in the Pecos River Valley of New Mexico and Texas. These data are essential for stewardship of aquatic resources in the Chihuahuan Desert ecoregion.

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DECLINING STATUS OF FRESHWATER MUSSELS IN THE RIO GRANDE, WITH COMMENTS ON OTHER BIVALVES

Robert G. Howells

ABSTRACT

Freshwater mussels (Family Unionidae) are one of the fastest declining faunal groups in North America due, in part, to their sensitivity to environmental degradation and modification. There has been a noteworthy paucity of data on unionid assemblages throughout the Rio Grande drainage despite its being a unique region of zoogeographic overlap between southern and northern faunas. Therefore, selected historic collections, state surveys by Texas Parks and Wildlife (TPW) 1992-1997, and federally-funded work by TPW and New Mexico Department of Game and Fish 1998-2001 were reviewed to provide a better understanding of mussel status in the system. At least 16 species of unionids occurred in the Rio Grande drainage of Texas, New Mexico, and Mexico. All have been dramatically reduced in both abundance and distribution in recent decades. Only six native species have been found alive

within the past ten years along with two others that are apparent introductions. Among taxa endemic to the Rio Grande, Salina mucket (*Potamilus metnecktayi*) and Mexican fawnsfoot (*Truncilla cognata*) have not been found alive since 1972 (though recently dead valves of the former collected in the late 1990s suggest it may still survive) and living or recently dead Rio Grande monkeyface (*Quadrula couchiana*) were last documented in 1898. The remaining taxa, including some that may be abundant elsewhere, also appear to have disappeared from the system over the past 10 to 100 years. Factors contributing to this decline include natural and anthropogenic desertification, water and land management practices, habitat modification, pollution, siltation, and increased salinity (in some areas). Unfortunately, there is little indication status of unionids in the Rio Grande will improve in the future.

INTRODUCTION

The Rio Grande of the United States and Mexico is one of the longest rivers in North America and is an area of overlap for many northern and southern faunal groups (Neck, 1982; Neck and Metcalf, 1988). Despite the political, ecological, and economical importance of the area, relatively little effort had been directed at understanding freshwater mussels (Family Unionidae) of the Rio Grande (Neck and Metcalf, 1988). Freshwater mussels are one of the most rapidly declining faunal groups in North America (Neves, 1993; Williams et al., 1993) including within the Rio Grande (Howells and Garrett, 1995; Howells and Ansley, 1999). Their sensitivity to environmental disturbance and degradation in conjunction with development of the region now places them at the pinnacle of ecological concern.

Historical information comes from Dall (1896) and others who reported on early boundary surveys. Strecker (1931), Neck and Metcalf (1988), Taylor (1967, 1997), and Johnson (1998) discussed these and other historic museum collections from the drainage. Cockerell (1902) reported finding unionids in the Pecos River drainage of New Mexico; Metcalf (1982) discussed recent, subfossil, and fossil records in the drainage basin; Metcalf (1974) commented on the Pecos River fauna; Metcalf and Stern (1976) briefly mentioned Rio Grande fauna; Neck (1984) commented on declining mollusks in Texas; and Neck and Metcalf (1988) reviewed mussels of the lower Rio Grande. Metcalf and Smartt (1972) discussed introduced mollusks. Several studies examined benthic invertebrates in general (Metcalf, undated, 1966; Bane and Lind,

1978; Davis, 1980a, 1980b, 1980c). Nonetheless, data from all these sources on the unionids of the Rio Grande was limited.

Recently, Texas Parks and Wildlife's (TPW) Heart of the Hills Fisheries Science Center (HOH) initiated statewide surveys of this group, including sites within the Rio Grande drainage (Howells, 1994a, 1994b, 1995, 1996a, 1996b, 1996c, 1997a, 1997b, 1997c, 1998a, 1998b; Howells and Garrett, 1995; Howells et al., 1996, 1997). In 1998, federally-funded work was jointly initiated by TPW and New Mexico Department of

Game and Fish to further examine the mussel fauna of the system (Howells, 1999, 2000, 2001a; Howells and Ansley, 1999; Lang, 2000). Johnson (1999) provided an analysis of the unionid fauna of the system, based on his interpretation of museum specimens and the published literature. Howells (2001b) summarized previous records and recent TPW surveys. Collectively, from these sources, a picture of the unionid faunal composition, abundance, and distribution within the Rio Grande Basin has started to emerge. This paper presents a condensed summary of previously published reports and surveys.

MATERIALS AND METHODS

Standard mussel sampling protocols employed in HOH work were presented in Howells (1995, 1996a). Specimen counting methods follow Howells (1995, 1996a) and include single valves and matched pairs of valves as one individual, two unmatched valves (one valve from each of two different animals) as two individuals, etc. (note that a valve or half of a shell originated from a single animal). Shell condition terminology follows (Howells, 1995, 1996a) and is presented in Table 1. Taxonomy follows Howells et al. (1996) and Turgeon et al. (1998), except for the common name for *Disconaias conchos* presented here. Details relating to specific sampling and collection sites were presented in Howells (2001b) as were maps to all locations.

Physicochemical aspects of the Rio Grande within Texas were examined by reviewing selected

water quality records of the U.S. Geological Survey for water years: 1968 (USGS, 1968), 1975 (USGS, 1975), 1986 (Buckner et al., 1986), 1996 (Gandara et al., 1996), 1999 (Gandara et al., 1999), and 2000 (Gandara et al., 2001). Data reported from these sources were averaged by site for all years combined, years for all sites combined, or both in an effort to identify temporal and upstream-to-downstream patterns. Precipitation data were obtained from the Texas Office of State Climatologist, College Station, Texas, and monthly and yearly totals were summed and decade (or partial decade) averages obtained.

Unless otherwise stated, recent survey work was largely confined to U.S. and boundary waters. Current status of bivalves in tributaries of the Rio Grande or other drainage basins in Mexico therefore remains unknown.

RESULTS AND DISCUSSION

SPECIES ACCOUNTS

At least 16 taxa of unionids occurred in the Rio Grande drainage of Texas, New Mexico, and Mexico (Howells et al., 1996). Three species are endemic to this drainage basin. Among the freshwater mussels, two are apparently recent introductions, as are Asian clams (*Corbicula* spp.; Family Corbiculidae), and another may have been lost, then reintroduced. Additionally, at least four fingernail clams (Family Sphaeriidae) have also been reported in the system.

Tampico pearlymussel (*Cyrtonaias tampicoensis*)

The native range of Tampico pearlymussel extends from the Brazos River, Texas, (Howells et al., 1996) south to at least the Río Papalopan system, Vera Cruz, Mexico (Johnson, 1999). Upstream records extend to the eastern boundary of Big Bend National Park (Howells, 1994b), with subfossil and fossil remains in the Pecos River upstream to Chaves and Eddy

Table 1. Shell condition terminology as presented by Howells (1995, 1996a). Note that it is not usually possible to determine exactly how long a freshwater mussel shell has been dead and many variables can impact specimen condition and rate of disintegration. Nonetheless, it is often useful to apply qualitative terms to help estimate time-since-death.

Term	Definition
Very recently dead	Soft tissue remains attached to the shell or valve; shell is in good condition, essentially as it would be in a living specimen; internal and external colors are not faded.
Recently dead	No soft tissue remains, but shell otherwise in good condition (looking like a living specimen that had been killed and cleaned); internally nacre is glossy and without evidence of algal or other staining, calcium deposition, or external erosive effects; internal and external colors are not significantly faded.
Relatively-recently dead	Shell is in good condition, but internally nacre is losing its glossy nature; algal or other staining, calcium deposition, or external erosive effects (or some combination of these) is evident on the nacre; internal and external colors are often somewhat faded.
Relatively-long dead	Similar to above, but more pronounced; shell epidermis often has sections beginning to flake.
Long dead	Shell shows signs of internal and external erosion, staining, calcium deposition, or some combination of these; most or all of the internal coloration and glossy nature of the nacre has faded (especially in species with colored nacre); shell epidermis often has major sections absent, or if present, clearly aged and flaking.
Very-long dead	Shell shows significant signs of internal and external erosion, staining, and calcium deposition more widely pronounced than above; coloration is often faded white or nearly so; often with relatively little epidermis remaining; for specimens in particularly erosive environments (scoured substrates, low pH waters, etc.), internal (e.g., shell teeth) and external features (e.g., pustules, ridges) are often weathered and smoothed, or otherwise exfoliated; shells are often chalky, brittle, and crumbling.
Subfossil	Shells have little or no epidermis remaining; nacre is faded white and entire shell is often bleached white; shell sometimes shows signs of erosion, staining, or calcium deposition of recent origin; shells are typically chalky and powdery to the touch, and often brittle and crumbling.

counties, New Mexico (Metcalf, 1982). It appears to be the most abundant unionid remaining in the Rio Grande. In the 1990s, populations were found in Amistad, Falcon, and Casa Blanca reservoirs, and in resacas, canals, oxbows, and reservoirs in Hidalgo and Cameron counties, Texas. However, the species is now apparently absent from areas upstream of Amistad Reservoir and Texas tributaries downstream to Starr County. Drought conditions, starting in mid-1995 and continuing into 1996 and beyond, also eliminated large numbers from Falcon and Amistad reservoirs as water levels fell.

Conchos disk (*Disconaias conchos*)

This species was first described by Taylor (1997) from 1969 collections near Saucillo, Rosetilla, and Julimes in the Río Conchos, Chihuahua, and a

1937 collection at Villa Juarez in the Río Sabinas, Coahuila, Mexico. It is apparently endemic to those systems and not known from the main stem of the Rio Grande or other tributaries. During desert fishes surveys by the HOH staff in 1994, 10 unmatched valves were found in the Río Conchos near Julimes, but were listed as unidentified (Howells and Garrett, 1995; Howells, 1996a). Among these, all were long dead except a single juvenile that was relatively-recently dead. Status of this species is uncertain, but the 1994 juvenile is the only suggestion the species may still survive.

Yellow sandshell (*Lampsilis teres*)

Yellow sandshell ranges throughout much of the central U.S. and is often common elsewhere in Texas (Howells et al., 1996). In the Rio Grande drainage, it

has been documented in the Pecos River, Val Verde County, Texas (Johnson, 1999); the Rio Grande near Del Rio, Val Verde County, downstream to Cameron County, Texas (Strecker, 1931); and the Río Sabinas, Coahuila (Johnson, 1999); and Río Salado near Anahuac, Nuevo Leon (Metcalf, 1982), Mexico. Neck and Metcalf (1988) described it as common in resacas in the Lower Rio Grande Valley and second in abundance only to Tampico pearlymussel. However, HOH surveys found only: 1) one subfossil valve in the Rio Grande downstream of Falcon Dam, Starr County, 1994 (Howells, 1996a); 2) a very-long dead valve in an oxbow pond, La Coma Tract – Lower Rio Grande National Wildlife Refuge, Hidalgo County, 1999 (Howells, 2000); and 3) a recently dead specimen in Elm Creek, Eagle Pass, Maverick County, 1992 (Howells, 1994a). A Late Archaic archeological site at Eagle Pass examined in 1995 (dated to 2,200-1,200 years before present) revealed yellow sandshell to have been the most abundant unionid material recovered (Howells, 1998c). Unfortunately, when the Elm Creek site was reexamined in 1996, the stream bottom was covered over 1 m deep in soft silt that had eliminated all unionids (Howells, 1997a). Despite a history of wide distribution in the system, the 1992 Elm Creek specimen was the only suggestion living animals might remain, but none have been located since.

Washboard (*Megalonaias nervosa*)

The range of this large waterbody species includes the central United States and lower river reaches in Texas from the Red River to the Rio Grande (Howells et al., 1996). In the Rio Grande, fossil shells have been found as far upstream as the Pecos River, Eddy County, New Mexico (Metcalf, 1982). Records in Texas range from a fragment at Elm Creek, Maverick County (Metcalf, 1982) and Las Moras Creek, Kinney County (Strecker, 1931) to fresh shells in the Rio Grande at Chapeno downstream of Falcon Dam in the mid-1970s (Neck and Metcalf, 1988). Records from Mexico include the Río Sabinas (Johnson, 1999) and fossil material from the Río Salado near Anahuac, Nuevo Leon, and subfossil specimens near Villa Juarez, Coahuila (Metcalf, 1992). Neck and Metcalf (1988) indicated washboard was rare in the Rio Grande and possibly extinct. Collections by HOH produced only a single very-long dead shell near Chapeno. However, D. Kumpe (South Padre Inland, Texas; pers. comm.) reported being given a fresh specimen taken from the

lower Rio Grande and seeing washboard shell fragments in dredge spoils at the mouth of the Edinburg Canal, Hidalgo County, Texas, in the mid-1990s. Additionally, P.D. Hartfield (U.S. Fish and Wildlife Service, Jackson, Mississippi; pers. comm.) interviewed a commercial musseler in the 1990s who was in possession of washboards he claimed had been recently collected in the lower Rio Grande Valley of Texas. The collection also contained Tampico pearlymussels (that could only have come from Texas or Mexico). HOH collections in Starr, Hidalgo, and Cameron counties failed to find fresh or living washboards, but some deepwater areas of the main channel remain to be examined, as does the main stem run between Amistad Dam and the Falcon Reservoir.

Texas hornshell (*Popenaias popeii*)

Though sometimes reported as endemic to the Rio Grande (Howells et al., 1997), Texas hornshell historically ranged south along the Mexican coastal systems at least to the Río Cazonas, Vera Cruz (Johnson, 1999). It has been found upstream in the Rio Grande to sites just downstream of Big Bend, Brewster County, Texas (Howells, 1999); upstream in the Pecos River to Chaves and Eddy counties, New Mexico (Cockerell, 1902; Metcalf, 1982); in the Devils River upstream of Dolan Springs, Val Verde County, Texas (Howells, 2000); as well as several Mexican tributaries of the lower Rio Grande (Johnson, 1999). A shell found in the Llano River in 1971 at Castell, Llano County (Ohio State University Museum collection), and another recently dead shell taken in the South Concho River in 1992 at Cristoval, Tom Green County (N. Strenth, Angelo State University, San Angelo, Texas; pers. comm.), both in the Colorado River drainage of Texas, are enigmatic. However, numerous collections throughout this system have failed to find evidence of other specimens. The only records in the past decade of living or recently dead specimens include one recently dead shell in 1992 from the mouth of San Francisco Creek, Brewster and Terrell counties, Texas (Howells, 1994a); three recently dead shells in 1998 found in the Rio Grande between Big Bend and San Francisco Creek, Brewster County, Texas (Howells, 1999); and an extant population in the Black River (Pecos River tributary), Eddy County, New Mexico (Lang, 2000). The Black River population in New Mexico and a possible relict population in the Rio Grande downstream of Big Bend may be the only surviving Texas hornshell in the United States.

Salina Mucket (*Potamilus metnecktayi*)

Salina mucket has been included under both *Disconaias* and *Potamilus* (Howells et al., 1996). Johnson (1998) redescribed endemic animals from the Rio Grande as *Potamilus metnecktayi* and seems to have combined a number of other related species to the south of the Rio Grande under *Disconaias disca*. In the Rio Grande, the species has been documented from just upstream of Boquillas Canyon, Brewster County, Texas (Howells, 1999), and from fossil material from the Pecos River, Eddy County, New Mexico (Metcalf, 1982), downriver to below Falcon Dam, Starr County, Texas (Neck and Metcalf, 1988), and the Río Salado of Tamaulipas, Coahuila, and Nuevo Leon, Mexico (Johnson, 1999). Living specimens were found in 1968 in the lower Pecos River, Val Verde County, Texas (Johnson, 1999) and fresh shells were found in the Rio Grande west of Del Rio, Val Verde County, in 1972 (Metcalf, 1982). No living specimens have been documented since. Two recently dead specimens were found at Dryden Crossing, Terrell County, Texas, in 1992 (Howells, 1994a); 11 recently dead and two relatively recently dead shells or valves were collected in 1998 between Dean Canyon, Brewster County, and 4.4 km downstream of San Francisco Creek, Terrell County (Howells, 1999); and a single relatively recently dead valve was found just upstream of Boquillas Canyon in 1999 (Howells, 2000), all in the Rio Grande. In nearly 30 years, these are the only records that suggest the Salina mucket is still extant.

Bleufer (*Potamilus purpuratus*)

The native range of bleufer extends from the Guadalupe-San Antonio drainage of Texas north and east (Howells et al., 1996). Previous reports of this species in the Rio Grande represent misidentified Tampico pearlymussels (Howells, 1997c). However, shells and living specimens were found in Amistad Reservoir, Val Verde County, Texas, in 1994 and 1995 where it appears to have been introduced (Howells, 1997c). It was documented there again in 1998 (Howells, 1999).

Giant floater (*Pyganodon grandis*)

Though widely distributed and often common in Texas and elsewhere in the United States (Howells et

al., 1996), there have been few reports of giant floater from the Rio Grande. Records include the El Toro Cement Agency Lake, El Paso County, Texas, 1969 (specimens at the U.S. National Museum and Philadelphia Academy of Science); Granjeno Lake (Strecker, 1931) and canals at Mercedes (Ellis et al., 1930), both Hidalgo County, Texas; and fossil material from Billingslea Draw near Toyah in the Pecos River drainage, Reeves County, Texas (Metcalf, 1982). None were taken in any HOH surveys conducted since 1992; however, in 2000, Lang (2000) found living specimens in the Pecos River upstream of Avalon Reservoir and in the main stem at Flume Park Carlsbad, Eddy County, New Mexico. Metcalf and Smartt (1972) considered giant floater to have been introduced at the El Paso site, as did Lang of the New Mexico specimens. The El Paso site has not been reexamined in many years and the status of that population is undetermined. Other than the population in New Mexico (possibly non-native stock), there is no other recent confirmation of the species in the Rio Grande drainage.

Southern mapleleaf (*Quadrula apiculata*)

Although southern mapleleaf is common throughout most of Texas and other Gulf states (Howells et al., 1996), it is absent from the fossil record (Metcalf, 1982), historical, and Mexican collections from the Rio Grande. Neck and Metcalf (1988) suggested it may be an introduction. It is known to occur in Lake Casa Blanca, Maverick County; Falcon Reservoir, Zapata County; and in the Rio Grande, canals, resacas, oxbows, and impoundments in Starr, Hidalgo, and Cameron counties, Texas (Neck and Metcalf, 1988; Howells, 1996a, 1997a, 1999, 2000).

Rio Grande monkeyface (*Quadrula couchiana*)

This endemic species has been reported as fossil remains as far upstream in the Pecos River as Eddy County, New Mexico, and the Pecos River near the mouth of Hackberry Draw, Ward County, Texas (Metcalf, 1982), with other material from the Devils River (Johnson, 1999) and downstream in the Rio Grande tributary of Las Moras Creek, Fort Clark (Brackettville), Kinney County, Texas (Strecker, 1931). Other subfossil and fossil specimens have been found at sites in the Río Salado, Coahuila and Nuevo Leon,

Mexico (Metcalf, 1982). A record from the Nueces River drainage, Zavalla County, Texas, is thought to be spurious (Johnson, 1999). This species may have been last seen alive or recently dead in the 1898 collections in Las Moras Creek (see Taylor, 1967). Neck (1984) reported D.W. Taylor had told A.L. Metcalf that he had found living specimens in the Río Conchos, Chihuahua, Mexico; however, no subsequent published record has been presented and none were found there during HOH surveys of this system in the 1990s. Otherwise, only fossil material has been found in the past century and the species may well be extinct.

False spike (*Quincuncina mitchelli*)

This species is known only from two disjunct populations, one in Central Texas and the second in the Rio Grande drainage (Howells et al., 1996). Living specimens have not been found in Central Texas since the late 1970s (Howells et al., 1997). Nearly all records from the Rio Grande are those of Metcalf (1982) who documented subfossil and fossil specimens from the Pecos River, Eddy County, New Mexico; the Pecos River drainage of Reeves and Ward counties, Texas; and in Mexico in the Río San Juan of Nuevo Leon, and Río Salado, Nuevo Leon and Coahuila, Mexico. Johnson (1999) noted an additional collection from the Río Salado, Tamaulipas, in the University of Michigan Museum of Zoology, but did not comment on the condition of the specimen. Surveys by HOH have failed to find even fossil fragments of false spike in the Rio Grande and the species is likely extinct in the basin. Two recently dead valves found in the lower Guadalupe River drainage in 2000 (Howells, 2001) are the only suggestion the Central Texas population has not been lost as well.

Lilliput (*Toxolasma parvus*)

Taxonomic confusion among species in this genus has been problematic, particularly with historic collections. Lilliput occurs throughout most of Texas and much of the United States (Howells et al., 1996), but in the Rio Grande, it is restricted to the lower reaches where it has been rather rare. Reports include Delta Reservoir, Hidalgo County (H.D. Murray; collection now at the Philadelphia Academy of Science) and resacas of the lower Rio Grande (Neck and Metcalf,

1988). However, during HOH surveys collections included only a single relatively recently dead valve in 1994 from Lake Casa Blanca, Maverick County (Howells, 1996a), and four living specimens in 1995 from Falcon Reservoir, Zapata County (Howells, 1996b). No other occurrences have been documented since.

Texas lilliput (*Toxolasma texasiensis*)

Although Turgeon et al. (1998) and Williams et al. (1993) recognized lilliput as well as Texas lilliput and western lilliput (*T. mearnsi*), genetic analysis (Howells, 1997b) failed to distinguish the latter taxon as a distinct species (Howells, 1997b). Howells et al. (1996) included western lilliput under Texas lilliput. Often locally abundant in much of Texas (Howells et al., 1996), Texas lilliput has only rarely been reported in the Rio Grande. Historic collections, all in Texas, include the mouth of the Devils River, Val Verde County, and Las Moras Creek, Fort Clark (Brackettville), Kinney County (Strecker, 1931; Taylor, 1967). Texas lilliput was third in abundance in archeological excavations along Elm Creek, Maverick County (Howells, 1998c), where its sexually dimorphic shells were readily apparent. However, it was not found in HOH survey sites throughout the basin examined during the past decade.

Mexican fawnsfoot (*Truncilla cognata*)

This Rio Grande endemic is known only from a small number of specimens. Originally described from the Río Salado, Nuevo Leon, Mexico in 1860, it has also been taken from the same river in the state of Tamaulipas (U.S. National Museum; Johnson, 1999). Metcalf (1982) reported a specimen of probably fossil origin from the Río Salado, Nuevo Leon. Metcalf (1982) also noted finding fresh shells in 1968 in the lower Pecos River and in 1972 in the Rio Grande west of Del Rio, both in Val Verde County, Texas. C.M. Mather (University of Science and Arts of Oklahoma, Chickasha; pers. comm.) collected a weathered valve in the Rio Grande 72 km west of Laredo in 1975. A relatively-long dead and deformed valve found in Falcon Reservoir in 1996 and initially attributed to this species (Howells, 1997b) may be a misidentification. None have been documented otherwise since the 1972 Metcalf collection.

Pondhorns (sp.?)*(Uniomerus tetralasmus – U. declivis)*

A pondhorn, given as *Unio manibius*, was taken in an early boundary survey at Río Agualeguas, Punttiagudo, near General Trevino, Nuevo Leon, Mexico, in the 1850s (see photograph in Johnson, 1999). Morrison (1976) synonymized *Unio minibus* and *Uniomerus declivis*, and he and Frierson (1903) both considered *U. tetralasmus* and *U. declivis* to be valid species. However, Johnson (1999) combined them. Neck (1987) considered both the Río Agualeguas specimen and others from Baffin Bay drainage, the next system to the north, to be *U. tetralasmus*. Many Texas *Uniomerus* populations are intermediate between the two species (HOH unpublished data), as are both Río Agualeguas and Baffin Bay examples, and their true taxonomic affinities remain unclear. Neck and Metcalf (1988) indicated specimens collected in the 1920s in Cottingham Resaca, Brownsville, Texas, were present in the Corpus Christi Museum collection; however, this collection could not be located for examination. Regardless, Neck (1987) concluded that no members of this genus were currently known from the lower Rio Grande Valley. Recent work supports the conclusion that all pondhorns of any species appear to have been eliminated from the system.

Paper pondshell (*Utterbackia imbecillis*)

Paper pondshell is often common in most Texas drainages and elsewhere in the United States. In the Rio Grande, it has been documented from Matamoros in the mid-1800s, Tamaulipas, and Río Salado near Anahuac, Nuevo Leon, Mexico (Johnson, 1999). Historic sites in Texas include: Brownsville, Cameron County (Johnson, 1999); San Lorenzo Creek, Webb County (Johnson, 1999); Las Moras Creek, Kinney County (Strecker 1931); Rio Grande upstream of the Río Conchos in 1979, Presidio County (Metcalf, undated); and Beaver Lake on the upper Devils River, Val Verde County (Strecker, 1931). More recent Texas records include: an oxbow pond and Sapo Lake, Hidalgo County (Howells, 2000); Falcon Reservoir, Zapata County (Howells, 1998a, 1999); Lake Casa Blanca, Maverick County (Howells, 1994a, 1996a, 1997a); the lower Devils River, Val Verde County (Howells, 1996a); and Lake Balmorhea, Reeves County (Howells, 1998a, 1999). Two lots of paper pondshell taken by R.D. Camp early in the past century upstream of El Paso in the San Jose River near San Rafael, Valencia County, New Mexico are present in the Corpus Christi Museum collection (J. Deisler-Seno, Corpus Christi Museum; pers. comm.). Small populations probably persist in ponds, impoundments, and backwaters from the lower Rio Grande Valley upstream to the lower Devils River and in spring runs at Balmorhea of the Pecos River drainage. Status of the species in Mexico and New Mexico is uncertain.

SPECIES REPORTS OF DOUBTFUL VALIDITY

Threeridge (*Amblema plicata*) was reported by Dall (1896), as *Unio undulatus*, from Kinney County based on a single “badly broken, and much worn right valve.” In as much as even fossil specimens are otherwise lacking from the system, it seems likely the valve was either a misidentified washboard or the collection locality was incorrect. Round pearlshell (*Glebula rotundata*) was listed by Simpson (1914) and Strecker (1931), based on an earlier report by Conrad (1855), from the Rio Grande. The species is otherwise not known from areas south of Green Lake at

the mouth of the Guadalupe River and reports of round pearlshell from the Rio Grande are likely misidentified Tampico pearlymussel (Howells et al., 1996). Although it is probably not part of the Rio Grande assemblage, P.D. Hartfield (U.S. Fish and Wildlife Service, Jackson, Mississippi; pers. comm.) reported seeing this species among others a commercial musseler claimed to have taken from the Rio Grande (as discussed above). Nonetheless, recent survey efforts failed to find it.

FRESHWATER BIVALVES IN OTHER FAMILIES

Asian clam (*Corbicula fluminea*), first found near El Paso in the mid-1960s (Metcalf, 1966), occurs throughout the drainage basin (Howells et al., 1996). A second possible species in the genus has also been reported from the Rio Grande (Hillis and Patton, 1981; Hillis and Mayden, 1985). Among the fingernail clams, several have been documented including: long

fingernailclam *Musculium transversum*, ubiquitous peaclam *Pisidium casertanum*, striated fingernailclam *Sphaerium striatinum*, mottled fingernailclam *Eupera cubensis* (Metcalf, undated; Davis, 1980a, b, c). Asian clam is an undesirable exotic, but its status in the Rio Grande has rarely been quantified. Fingernail clams are even less well studied than unionids.

POSSIBLE REASONS FOR DECLINE

CLIMATE AND WEATHER

The Rio Grande drainage basin has been naturally changing from a cooler, more-moist climate to a warmer and dryer situation. Wilkins (1992) described a general climatic trend toward warming and drying since the end of the Wisconsin glaciation about 18,000 years ago. Smith and Miller (1986) discussed a regional shift from woodlands to grasslands and deserts from about 11,500 years ago. Loss of species like washboard from upriver areas in New Mexico and its range reduction to the lower-most Rio Grande reflects this ongoing climatic change. Similarly, Rio Grande monkeyface and false spike may have been in a mode of natural decline prior to European impact.

Recent precipitation pattern changes may also play a role in mussel decline. Long-term precipitation

records for selected sites in Texas yield 10-year averages demonstrating a pattern of increasing rainfall at Brownsville (61.7 cm in the 1950s to 83.0 cm in the 1990s) and Laredo (112.0 cm in the 1940s to 133.6 cm in the 1990s). However, this pattern was less evident at Del Rio and was not found at El Paso. In addition to increasing net rainfall in some areas, weather records also indicate a shift to fewer light and moderate showers and more severe storm events (MICRA, 1995). Thus, recent decades have experienced more major storms producing scouring floods that directly destroy unionids and alter habitat, with long periods of insufficient precipitation in between resulting in dewatering losses. Freshwater mussels need stable conditions to survive and prosper. Current precipitation trends are clearly unfavorable.

WATER FLOW PATTERNS

Human impact appears to be the major reason for the massive reduction in mussel faunal diversity and abundance in the Rio Grande. Overgrazing, land clearing, construction of impervious surfaces, and other anthropogenic modifications have also contributed to increased runoff during storms and additional scouring and riverbed modifications. Increased groundwater pumping, in turn, reduces spring flows and subsequently river water levels during dry periods. All of these factors can contribute to negative impacts on unionids.

Flow rates reflect an interrelationship of both precipitation and water retention or releases from impoundments, as well as other factors. The U.S. Na-

tional Park Service (NPS, 2001) reported 69-86% of the water in the Rio Grande downstream from Presidio originated from the Río Conchos, Mexico, and that although a treaty between the U.S. and Mexico defined allotments related to annual flows, the treaty did not specify release schedules for Mexican rivers. Mean annual flows by site are least at El Paso and Brownsville, but greatest between Langtry and Rio Grande City (Table 3). Mean annual flows during the years examined display a pattern of decrease over time from the Rio Grande downstream of Amistad dam to Brownsville (Table 3). Flows at Laredo in 1975 averaged 149 m³/s, but dropped in 2000 to 43.2 m³/s. At Brownsville, flows were 100 m³/s in 1975, but fell to 1 m³/s in 1999 and 5 m³/s in 2000. In 2001, flow downstream of

Brownsville stopped and freshwater failed to reach the Gulf of Mexico at times. Historic droughts have also reduced or eliminated flows in the Rio Grande as in 1952, 1955, 1957, and 1958 when the riverbed was dry at Johnson Ranch, Big Bend National Park (NPS, 2001). Past dewatering likely reduced or eliminated

some unionid populations and the current pattern of flow rate decline poses an increasing threat again. However, large-volume releases below mainstem dams, which may actually occur during drought conditions, can cause scouring-related damage to mussels and aquatic habitat at downstream sites as well.

NUTRIENT LOADS

Mean annual levels of total phosphorus between El Paso and Brownsville and in the Pecos River at Langtry were below 0.5 mg/L (Table 2). Similarly, sulfate ranged from 161-290 mg/L in the main channel of the Rio Grande, was low in the Devils River (9 mg/L), but averaged over 2,000 mg/L in the upper and central Pecos River of Texas (Table 2). Among all years combined, nitrate levels averaged below 1.4 mg/L in the main channel and the Pecos River at Langtry, except at Brownsville where it slightly exceeded 1.6 mg/L (Table 2). Interestingly, nitrate concentrations

show a significant pattern of decline over time. In 1968, levels were very high in the Rio Grande at Langtry (3.2 mg/L), Laredo (3.7 mg/L), and Brownsville (6.0 mg/L), but decreased to < 1.0 mg/L at these same sites in 2000, with a decline to 0.2 mg/L at Brownsville (Table 3). Although there is little published information associating nutrient levels and freshwater mussels, it seems unlikely that concentrations currently found in the Rio Grande would be problematic (but with some concern about sulfate in the upper and central Pecos River).

Table 2. Mean values for selected physicochemical parameters at locations on the Rio Grande, Pecos River, and Devils River, Texas, obtained from measurements presented in U.S. Geological Survey reports for water years 1968 (USGS, 1968), 1975 (USGS, 1975), 1986 (Buchner et al., 1986), 1996 (Gandara et al., 1996), 1999 (Gandara et al., 1999), and 2000 (Gandara et al., 2001). Values for Amistad, Falcon, and Anzalduas dams were actually taken in the Rio Grande downstream of those structures.

Parameter	Rio Grande – Main channel							
	El Paso	Langtry	Amistad Dam	Laredo	Falcon Dam	Rio Grande City	Anzalduas Dam	Brownsville
Flow rate (m ³ /s)	22.2	52.1	45.6	83.1	63.5	68.2	34.6	33.0
Total phosphorus (mg/L)	0.41	0.9	<0.01	0.13	0.06	-	-	0.15
Sulfate (mg/L)	289	290	214	161	200	209	233	233
Nitrate (mg/L)	0.89	1.35	0.22	1.14	0.40	-	-	1.62
Chloride (mg/L)	148	117	141	116	116	136	167	181
Conductivity (µS/cm)	1368	1214	1103	959	971	1065	1020	1322
Suspended sediments (mg/L)	535	2685	5	137	20	-	-	64
Dissolved solids (mg/L)	863	772	663	598	589	645	730	761
Turbidity (NTU)	140	775	1	53	8	-	-	27

Parameter	Pecos River			Devils River
	Orla	Girvin	Langtry	Comstock
Flow rate (m ³ /s)	2.1	1.0	9.5	6.4
Total phosphorus (mg/L)	-	-	0.04	-
Sulfate (mg/L)	2012	3477	467	9
Nitrate (mg/L)	-	-	0.92	0.94
Chloride (mg/L)	3477	5818	777	15
Conductivity (µS/cm)	12651	21585	34551	381
Suspended sediments (mg/L)	-	-	12	22
Dissolved solids (mg/L)	9742	12953	2038	214
Turbidity (NTU)	-	-	10	5

Table 3. Annual mean values for flow rate, nitrate concentration, and suspended sediments at locations on the Rio Grande, Pecos River, and Devils River, Texas, obtained from measurements presented in U.S. Geological Survey reports for water years 1968 (USGS, 1968), 1975 (USGS, 1975), 1986 (Buchner et al., 1986), 1996 (Gandara et al., 1996), 1999 (Gandara et al., 1999), and 2000 (Gandara et al., 2001). Values for Amistad, Falcon, and Anzalduas dams were actually taken in the Rio Grande downstream of those structures.

	Year					
	1968	1975	1986	1996	1999	2000
Flow Rate (m ³ /s)						
Rio Grande – El Paso	-	-	27.2	16.9	19.9	24.8
Rio Grande – Langtry	-	54.8	52.0	35.2	65.3	53.4
Rio Grande – Amistad Dam	-	-	80.4	46.2	30.4	25.5
Rio Grande – Laredo	-	148.9	110.9	46.8	65.8	43.2
Rio Grande – Falcon Dam	-	-	98.4	61.4	32.7	61.5
Rio Grande – Rio Grande City	-	-	97.4	64.0	56.6	54.9
Rio Grande – Los Ebanos	-	-	98.3	30.4	41.7	38.1
Rio Grande – Anzalduas Dam	-	-	39.0	33.4	34.8	31.1
Rio Grande – Brownsville	81.1	99.5	8.2	3.1	1.0	5.1
Pecos River – Orla	2.4	2.6	0.8	2.5	-	-
Pecos River – Langtry	0.5	2.0	7.2	16.0	0.6	0.7
Devils River – Comstock	-	-	6.0	6.9	3.6	4.0
Nitrate (mg/L)						
Rio Grande – Langtry	3.2	1.5	1.1	0.7	0.8	0.8
Rio Grande – Laredo	3.7	0.6	0.4	-	0.7	0.5
Rio Grande – Brownsville	6.0	0.3	0.9	0.1	0.4	0.2
Pecos River – Langtry	-	1.8	0.6	0.3	0.4	0.9
Devils River – Comstock	-	-	1.2	0.7	-	-
Suspended sediments (mg/L)						
Rio Grande – Langtry	-	-	654	2364	4113	3609
Rio Grande – Falcon Dam	-	-	122	29	19	20
Rio Grande – Brownsville	-	-	120	63	47	64

CHLORIDE AND CONDUCTIVITY

Chloride concentrations and associated conductivity values may have a direct impact on mussel presence or absence at some locations in the basin. Averages among years produce chloride values from El Paso to Brownsville of 116-181 mg/L and conductivity values of 959-1,368 μ S/cm that are typically greatest at up- and downstream sites (Table 2). In the Devils River, chloride averaged only 15 mg/L and conductivity 381 μ S/cm (Table 2). However, chloride concentrations in the upper (Orla), central (Girvin),

and lower (Langtry) Pecos River were dramatically elevated to 3,477, 5,818, and 777 mg/L, respectively (Table 2). Similarly conductivity at these same sites was 12,651, 21,585, and 34,551 μ S/cm, respectively (Table 2).

Thus the Pecos River is the major source of elevated chloride and conductivity values in the system. Natural salt seeps and deposits are present in the area, but groundwater pumping that has reduced freshwa-

ter input, long periods of reduced precipitation, and brines from oil and gas drilling operations likely contribute to current saline conditions. Chloride and conductivity levels in the mainstem Rio Grande are probably not limiting (though upper lethal limits for most unionids remain undefined). However, levels in the upper and central reaches of the Pecos River in Texas are probably sufficiently high as to preclude long-term

mussel survival. At most sites in the Pecos River, Texas, even disturbance-tolerant Asian clam is not present. In addition to the dam at Red Bluff Reservoir, salt waters probably genetically isolate the Texas hornshell population in the Black River, New Mexico, from any survivors in the Rio Grande downstream of Big Bend.

TOTAL DISSOLVED SOLIDS AND SEDIMENT LOAD

Total dissolved solids average from El Paso to Brownsville range from 589-863 mg/L, are lower in the Devils River (214 mg/L), but dramatically elevated in the Pecos River (2,038-12,953 mg/L)(Table 2). Mean suspended sediment levels (mg/L) by location from upstream to downstream in the Rio Grande were 535 (El Paso), 2,685 (Langtry), 5 (Rio Grande downstream of Amistad Dam), 137 (Laredo), 20 (downstream of Falcon Dam), and 64 (Brownsville), and were 12 (Pecos River, Langtry) and 22 (Devils River, Comstock)(Table 2). The increase in suspended sediment agrees with field observations of heavy recent, silt deposition at and upstream (in the Rio Grande) of the Pecos River mouth. Indeed, comparing annual means indicates suspended sediment loads typical of the system in general in the Rio Grande at Langtry in 1986 (654 mg/L), but with levels elevating in the late 1990s to 2,364 (1996), 4,113 (1999), and 3,609 mg/L (2000)(Table 3). U.S. Geological Survey (Gandara et al., 2001) data from a sampling station in the Rio Grande just downstream of the Río Conchos indicated an average suspended sediment load in 2000 of 169

mg/L, lower than location averages elsewhere in the mainstem of the Rio Grande. The Río Conchos, then, appears not to be the main source of the suspended sediments that have increased between Presidio and Langtry. Turbidity measurements follow the same pattern found with suspended sediments (Table 2). Although the primary source of increased sediment is unclear from recent freshwater mussel survey work, certainly overgrazing, development, and other typical anthropogenic factors contribute. It is noteworthy the only areas in the Rio Grande of Texas where Texas hornshell and Salina mucket may still be surviving are in or just upstream of the major site of silt deposition. Deep soft silts are unacceptable mussel habitat (Howells et al., 1996). Finally, despite heavy silt loads and deposition at Langtry, Amistad Reservoir appears to block much of this material from progressing downstream, as does Falcon Reservoir to a lesser extent downriver. Although this may benefit mussels still surviving in the river below these dams, it suggests a questionable future for reservoir populations.

RESERVOIR CONSTRUCTION AND POLLUTION

Reservoir construction and resultant impounded waters may eliminate some mussels and their natural habitat while creating additional habitat for other species (Howells et al., 1996). Indeed, neither Texas hornshell nor any of the endemic unionids in the Rio Grande are known from reservoirs and may well require flowing water situations. Conversely, Tampico pearlymussel adapts well to reservoirs, despite having evolved in riverine environments (Howells, 1996c; Howells et al., 1996). Even when unionids are able to survive in impounded waters, water management prac-

tices can still be destructive. Rapidly fluctuating water levels and long periods of dewatering are common sources of mortalities. These problems were particularly evident in Amistad and Falcon reservoirs when water levels began to decline substantially in mid-1995. Reservoirs may also block movement of host fishes required for parasitic unionid larvae. Host fishes utilized by Tampico pearlymussel, yellow sandshell, washboard, bleufer, giant floater, lilliput, Texas lilliput, pondhorn (sp.?), and paper pondshell (Howells, 1997d; Howells et al., 1996), as well as Texas hornshell (Gor-

don, 2000) are probably not seriously limited by dams in the basin. However, hosts required by Conchos disk, Salina mucket, southern mapleleaf, Rio Grande monkeyface, and Mexican fawnsfoot are unknown; thus host availability issues cannot be addressed.

Pollution and habitat modification associated with human development have probably impacted unionid fauna of the Rio Grande as they have throughout the country, but documentation locally is largely lacking.

However, some areas reportedly have a high potential for chemical impacts (Kelly and Reed, 1998). Tremendous human population growth and human development has occurred in the drainage basin over the past 30 years (Kelly and Reed, 1998). The North American Free Trade Agreement (NAFTA) of recent years has no doubt enhanced the speed and extent of this development. It is not apparent, however, that any NAFTA-related development has ever specifically considered impacts on unionids.

COMMERCIAL HARVEST

Freshwater mussels have supported important commercial shell fisheries elsewhere in Texas and the U.S. (Howells, 1993). Historically, the U.S. military harvested local shells for buttons early in the past century in Cameron County, Texas (Neck, 1990), and a button factory operated briefly at Mercedes, Hidalgo County, Texas (Garrett, 1929). Pearl harvesters that seek Tampico pearlymussel in the Colorado and Brazos drainages, Texas (Howells, 1993, 1996c) appear not

to focus similar harvest efforts on the Rio Grande. A survey of both resident and non-resident mussel license holders in Texas (Howells, 1993) found no respondents indicating they took mussels from the Rio Grande. There is no indication that commercial harvest for shells or pearls is, or has ever been, economically important or contributed to the decline of the fauna locally.

SUMMARY

All unionid species in the Rio Grande drainage basin have been dramatically reduced in abundance and distribution both historically and in recent years. Among the 16 freshwater mussel taxa reported in the system, two are likely extinct (Rio Grande monkeyface and false spike). Seven other taxa have not been documented in recent years and are either extinct or reduced to very low numbers (Conchos disk, Mexican fawnsfoot, yellow sandshell, washboard, lilliput, Texas lilliput, and pondhorn (sp.?). Texas hornshell is still extant in a short stretch of the Black River in southern New Mexico and a small relict population may also be present in the Rio Grande downstream of Big Bend. Similarly, Salina mucket may still survive in low num-

bers just downriver of Big Bend. Tampico pearlymussel, paper pondshell, and introduced southern mapleleaf are currently maintaining populations in Texas waters, and a bleufer population introduced in Amistad Reservoir is presumed to be present as well. Among other bivalves, Asian clams are widely distributed and even abundant in some areas. Fingernail clams are still present, but current species, abundance, and distribution are poorly defined. Unfortunately, ecological sensitivity of unionids, projected future development within the drainage, and general disinterest among regulatory authorities suggest an extremely dim future for this unique faunal group.

ADDENDUM

Since this manuscript was drafted, unionid collections in 2002 in the Rio Grande in Webb County have included one living and several recently dead shells or valves each of washboard and Texas hornshell, as

well as several recently dead shells and valves of Mexican fawnsfoot. Additional survey efforts are planned to better define the status of these species in the Webb County area.

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FISH ASSEMBLAGES OF THE RÍO CONCHOS BASIN, MÉXICO, WITH EMPHASIS ON THEIR CONSERVATION AND STATUS

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ABSTRACT

The Chihuahuan Desert region contains a number of unique aquatic environments, but with few exceptions, these have been little studied. We sampled the Río Conchos Basin in 1994 and 1995 to assess the status of the fishes of this region. Most sites showed some degree of human-induced impacts. A number of potentially threatened fishes were either abundant at only a few sites or rare or absent throughout the lo-

calities sampled. Comparisons with collections taken during the 1950s indicate that the basic fish fauna is largely intact. However, there appears to be diminished relative abundances of "large river" forms in favor of non-natives (primarily an African cichlid, *Oreochromis aureus*) and "quiet-water" native fishes. This change seems related to decreased flows and regulated flow regimes from dams in the basin.

RESUMEN

La región del desierto de Chihuahuan contiene un número de ambientes acuáticos únicos, pero con pocas anomalías, éstos se han estudiado poco. Porque muchos de los pescados en la región se piensan para ser amenazados con la extinción o han ido extintos, muestreamos el lavabo de Río Conchos en 1994 y 1995 para evaluar el estatus de los pescados de esta región. La mayoría de los sitios mostraron un cierto grado de impactos humano-inducidos. Un número de pescados potencialmente amenazados eran o abundantes en

solamente algunos sitios o raro o ausente a través de los lugares muestreó. Las comparaciones de nuestros datos a las colecciones tomadas durante los años 50 indican que mientras que la fauna básica de los pescados en la región es en gran parte intacta, aparece ser reducciones en formas del "río grande" y aumenta las formas del ambiente introducido de los pescados y de la "reservado-agua". Estos cambios aparecen relacionados a disminuido y los regímenes regulados del flujo.

INTRODUCTION

The limited aquatic habitats of the Chihuahuan Desert have undergone substantial anthropogenic modification in the last hundred years, including reduced water quality, diversion of surface water, overdrafting of groundwater, channelization, impoundment, and introduction of non-native species (Miller and Chernoff, 1979; Propst and Stefferud, 1994; TNRC, 1994; IBWC, 1994; Lee and Wilson, 1997; Edwards et al., 2001). Impacts from these modifications are only now being documented and few baseline data exist concerning the ecological requirements for most of the aquatic species.

Approximately half of the native fishes of the Chihuahuan Desert are threatened with extinction or are extinct (Hubbs, 1990). Documented extinctions from the northern Chihuahuan Desert include Maravillas red shiner (*Cyprinella lutrensis blairi*), phantom shiner (*Notropis orca*), Rio Grande bluntnose shiner (*Notropis simus simus*), and Amistad gambusia (*Gambusia amistadensis*) (Miller et al., 1989; Hubbs, this volume). Some noteworthy extirpations include Rio Grande shiner (*Notropis jemezanus*) from the New Mexico portion of the Rio Grande (Propst et al., 1987) and Rio

Grande silvery minnow (*Hybognathus amarus*), Rio Grande cutthroat trout (*Oncorhynchus clarki virginalis*) and blotched gambusia (*Gambusia senilis*) in Texas (Bestgen and Platania, 1990, 1991; Hubbs et al., 1991) as well as others. The status of a number of fishes in the northern Chihuahuan Desert is poorly understood, particularly for the Mexican portion of their ranges. Endemic species other than fishes also are being lost from this area (Howells and Garrett, 1995; Howells, this volume; Lang et al., this volume).

In this paper, we present data on fish collections from 14 localities in the Río Conchos Basin in 1994 and 1995. We compare our results with a series of collections taken from the basin nearly 40 years earlier, and we comment on the status of several imperiled fishes for which the Río Conchos Basin represents a significant portion of their geographic distribution.

STUDY AREA

The Río Conchos receives its water from a series of tributaries originating in the Sierra Madre Occidental in Chihuahua, Mexico, along with a number of springs, seasonal rains and periodic tropical storms (Tamayo and West, 1964). The climate ranges from subhumid and temperate in the Sierra Madre Occidental to semiarid and warm in the central Plateau and warm and arid in its northern-most reaches; temperatures often exceed 40°C and precipitation averages about 315 mm/year, with greater amounts in the mountain areas and lesser amounts in the central and northern portions of the state (Kelly, 2001). The Mexican state of Chihuahua has over two million people, with 80% living in small to medium sized towns and the remainder centered in the cities of Juarez and Ciudad Chihuahua. Anthropogenic impacts range from deforestation and mining in the Sierra Madre, impoundments on both large and small streams throughout the region, manufacturing in the cities, and agriculture

throughout the central Plateau, especially surrounding Ciudad Chihuahua (Comisión Nacional del Agua, 1997). Only the largest cities have wastewater treatment plants and most rural areas lack even basic sewage collection and disinfection facilities (Kelly, 2001). Some major streams (for example, the Río Florido) are severely impacted with high levels of oil, fecal coliform bacteria, discharges from chemical plants, and pollution from agricultural return flows (Comisión Nacional del Agua, 1997). The Río Conchos is the primary source of water for the Rio Grande downstream of Presidio, Texas. The average annual flow of the Rio Grande immediately upstream of its confluence with the Río Conchos is approximately 72 thousand acre-feet ($8.9 \times 10^7 \text{ m}^3$), whereas the Río Conchos contributes an annual average of 779 thousand acre-feet ($9.6 \times 10^8 \text{ m}^3$), far exceeding the input from any other tributary of the Rio Grande System (IBWC, 1990; Eaton and Hurlburt, 1992; TNRCC, 1994).

MATERIALS AND METHODS

We collected fishes from 14 locations (combined into 11 stations) throughout the Río Conchos System of Chihuahua (Figure 1). The Río Parral was not sampled because locals advised us that the waters were too polluted to even safely wade in. Our station designations, specific sampling localities and sampling dates were as follows: Station 1, springs at Ojo Talamantes, 16 km NE of Allende, 6 August 1994; Station 2, the Río Conchos at Valle de Zaragosa, 6 August 1994; Station 3, Río San Pedro S of Satevó at Highway 24 crossing, 6 August 1994; Station 4, Río Santa Isabela, 20 km downstream from Riva Palacio, 7 August 1994;

Station 5, Río Santa Isabela immediately upstream from Riva Palacio, 7 August 1994; Station 6, Río Conchos, immediately downstream from Julimes, 5 August 1994; Station 7, a series of three collections at the springs and outflows at San Diego de Alcalá, 24 October 1995; Stations 8a and 8b, Río Chuviscar near San Diego de Alcalá, 4 August 1994 (Station 8a) and 24 October 1995 (Station 8b), respectively, during and after construction of a water pipe to draw water from the stream; Station 9, headwaters of Río Chuviscar, near Highway 160, approximately 15 km S of Namiquipa, 23 October 1995; Station 10, Río Conchos at Cuchillo

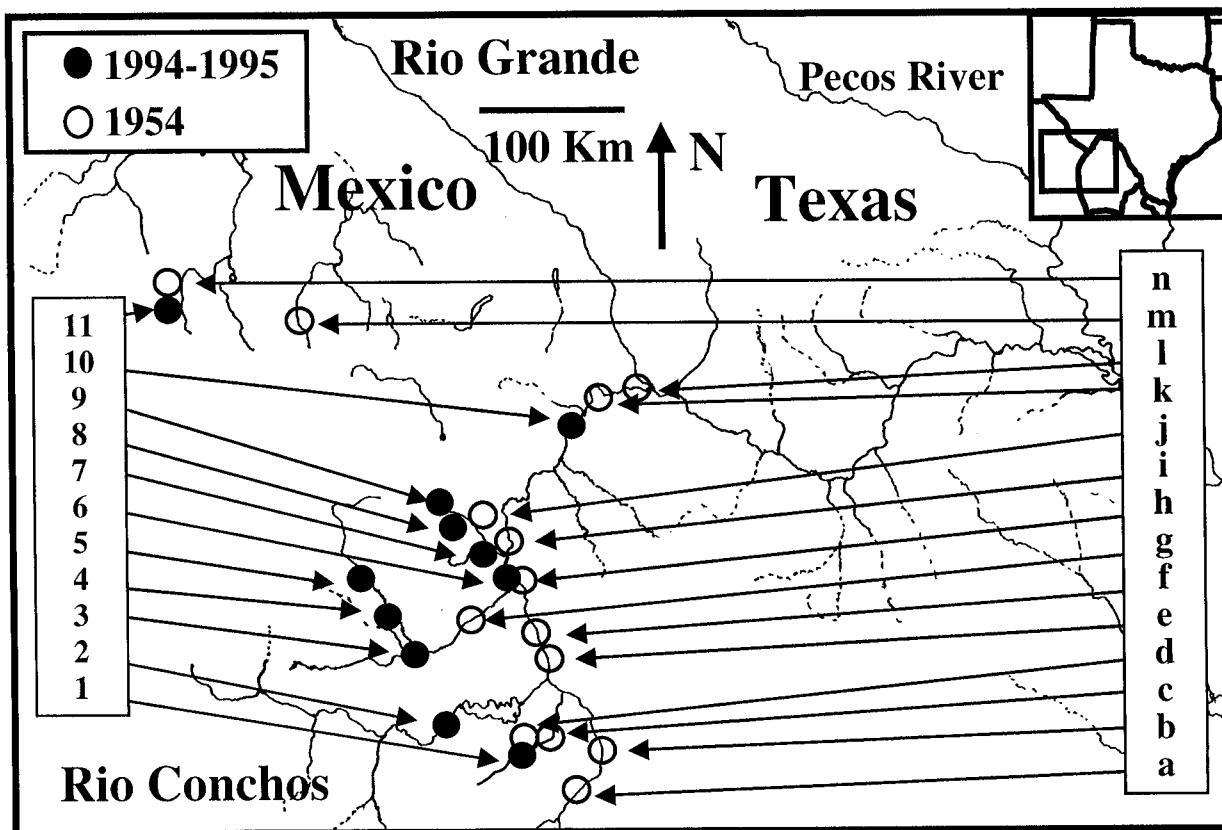


Figure 1. Map of Río Conchos study area. Numerals indicate most current collection sites and letters indicate comparative historical collection sites. See text for collections localities.

Parado, about 30 km from the confluence with the Río Grande, 3 August 1994; and, Station 11, springs at Ojo del Arrey, approximately 4 km NE of Angostura, 10 August 1994. The Río Florida was not sampled because it was dry or too seriously polluted throughout much of its course.

We intensively sampled contiguous segments of habitats with seemingly pristine conditions so as to represent, to the greatest degree possible, the natural biota. We selected multiple sampling sites at each location and collected at each site until we detected no additional changes in species occurrence and relative abundances in the sample. In general, all available microhabitats were sampled roughly in proportion to their occurrence. Sites were sampled with seines 3 m (3 mm mesh) to 10 m (6 mm mesh) long or electrofishing in all available habitats. At most sites waters were too shallow for effective electrofishing, and seining was the major method. At each location, all specimens collected were identified and enumer-

ated. A representative subsample of each species (except those not allowed by permit) was retained. Our data are presented as relative abundances to facilitate comparisons. We compared our data to museum collection records (Texas Natural History Collections, The University of Texas at Austin) of a series of collections taken by Clark Hubbs and Victor Springer in June and July 1954 and by Hubbs and Oscar Wiegand in December 1954 from similar areas (and using seines) throughout the Río Conchos basin. We collected in all available habitats within a fewer number of larger sites with generally three to six teams of collectors at each station, whereas the 1954 collectors sampled at a greater number of smaller habitats, in part, because only two people were sampling (C. Hubbs, University of Texas at Austin, pers. comm.). We combined a number of the 1954 sampling stations into a smaller series of stations in order to be more comparable to our collection data. The stations created for the 1954 comparisons (Figure 1) were as follows. Station a, Río Florida, at Highway 45 crossing, 17.6 km ESE of

Villa Ocampo, 27 June 1954; Río Florido at Espirito Santo, 8 km ESE of Villa Ocampo, 27 June 1954, a tributary of the Río Florido, 24.6 km ESE of Parral, 27 June 1954, and the Río Florido 0.8 km W of Villa Ocampo, 27 June 1954; Station b, Río Florido at Guadalupe, 22.4 km E of Parral, 26 June 1954; Station c, a series of collections at the springs and associated outflows near the Ojo de la Hacienda Dolores including the Río Valle de Allende, at Valle de Allende, 27 June 1954, El Ojo de la Hacienda Dolores, 8 km S of Jimenez and 3.2 km SE of Dolores, 30 June 1954, an irrigation ditch draining El Ojo de la Hacienda Dolores, 30 June 1954, El Ojo Almoloya, 3.2 km W of Estacion Troya, 30 June 1954, an irrigation ditch at Highway 45, 1.6 km SE of Bachimba, 1 July 1954, Río Parral, 3.2 km W of Parral and approximately 0.5 km W of the railroad bridge, 26 June 1954, Río Valle de Allende, 1.6 km W of Valle de Allende at the small dam, 30 December 1954, a ditch near El Ojo Almoloya, 2.4 km W of Troya, 31 December 1954 in an irrigation ditch near Ojo Hacienda Dolores at points 3.2 and 6.4 km S of Jimenez, 31 December 1954, El Ojo de La Hacienda Dolores, 8 km S of Jimenez and 3.2 km SE of Dolores, 31 December 1954, and Ojo Hacienda Dolores,

8 km S of Jimenez, 31 December 1954; Station d, Ojo de San Gregorio, 0.5 km W of San Gregorio and 19.2 km ENE of Parral, 31 December 1954; Station e, Río Conchos at Camargo, 25 June 1954, Río Conchos at La Cruz (several different sites), 25 June 1954, Río Conchos, 1.6 km N of Saucillo, 28 June 1954, Río Conchos at highway crossing, 19.2 km SW of Camargo, a tributary of the Río Conchos, 1.6 km E of San Francisco de Condios and 24 km SW of Camargo, both on 28 June 1954; Station f, Río Conchos at Saucillo on the E channel at ford, 25 June 1954; Station g, Río San Pedro, 1.6 km SW of Meoqui, 24 June 1954, Río San Pedro at Meoqui, 30 December 1954; Station h, Río Conchos at Julimes, 24 June 1954; Station j, was the Río San Pedro, 0.4 km SW of confluence with Río Conchos, 24 June 1954, Río Sacramento 1.6 km N of Ciudad Chihuahua, 1 July 1954; Station k, Río Conchos, 32 km W of Rio Grande confluence, 14 June 1954; Station l, Río Conchos, 1 km. upstream of confluence with Rio Grande, 13 June 1954, Río Conchos, 6.4 km W of Rio Grande confluence at Sierras Navas, 14 June 1954; Station m, Río del Carmen at El Carmen, 1 July 1954; Station n, Río Santa Maria at Buenaventura, 2 July 1954.

RESULTS

1994-1995 COLLECTIONS

Our collections yielded 18,371 specimens representing 37 species (Table 1). Streams within the Río Conchos Basin are characterized by a relatively large minnow (Cyprinidae) component that includes *Cyprinella lutrensis*, *Notropis chihuahua*, *Notropis braytoni*, *Notropis jemezianus*, *Campostoma ornatum*, *Macrhybopsis aestivalis*, and *Rhinichthys cataractae*. Other typical species were *Scartomyzon austrinus*, *Ictalurus lupus*, *Astyanax mexicanus*, *Cyprinodon eximius*, and *Gambusia senilis*. Several wide-ranging species were found, including *Dorosoma cepedianum*, *Pimephales promelas*, *Ictalurus furcatus*, *Pylodictis olivaris*, *Gambusia speciosa*, *Lepomis cyanellus*, *L. megalotis*, *L. macrochirus*, and *Micropterus salmoides*. As expected, mainstem localities contained more species and tributary streams contained a subset of the total species complement.

The springs at Ojo de Talamantes (Station 1) and those at Ojo del Arrey (Station 11) were quite different from the other localities sampled. At the former site, an undescribed species of *Gambusia* accounted for over 70% of the fishes captured, while at the latter site, *G. speciosa* and an undescribed species of *Cyprinodon* accounted for more than 95% of the fishes captured. Each of these springs has been substantially modified. The springs at Talamantes have been transformed into an aquatic tourist park and impounded with a low concrete dam and the springs at Ojo del Arrey have been developed into a swimming pool, with a small outflow spring run.

The headwaters of the Río Chuviscar (Station 9), the Río San Pedro (Station 3) and the Río Chuviscar at San Diego de Alcalá (Station 8b) all contained large

Table 1. (cont.)

Species	Station										
	a	b	c	cl	c2	d	e	f			
<i>Ictalurus punctatus</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Pylodictis olivaris</i>	—	—	—	—	—	—	0.33	—	—	—	—
<i>Cyprinodon eximius</i>	0.30	—	2.00	—	—	—	2.27	14.08	—	—	—
<i>Cyprinodon macrolepis</i>	—	—	—	5.44	—	—	—	—	—	—	—
<i>Cyprinodon sp.1</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Cyprinodon sp.2</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Fundulus zebrinus</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Gambusia speciosa</i>	—	—	—	—	—	120.16	—	—	—	—	—
<i>Gambusia alvarezi</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Gambusia hurtadoi</i>	—	—	—	91.48	—	—	—	—	—	—	—
<i>Gambusia senilis</i>	2.11	56.69	35.38	—	92.91	—	8.10	—	—	—	—
<i>Gambusia sp.1</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Gambusia sp.2</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Menidia beryllina</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Lepomis cyanellus</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Lepomis gulosus</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Lepomis macrochirus</i>	—	—	—	—	—	—	5.99	18.45	—	—	—
<i>Lepomis megalotis</i>	1.81	—	—	—	—	—	1.94	0.97	—	—	—
<i>Micropterus salmoides</i>	—	—	—	—	—	—	3.24	26.70	—	—	—
<i>Etheostoma australe/potisi</i>	—	—	0.71	—	—	—	0.16	—	—	—	—
<i>Aplodinotus grunniens</i>	—	—	—	—	—	—	—	—	—	—	—
<i>Oreochromis aureus</i>	—	—	—	—	—	—	—	—	—	—	—
Specimens captured	659	157	1,204	2,188	874	496	618	206	—	—	—
Number of species	15	5	10	5	4	1	17	8	—	—	—
Percent introduced	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	—	—	—

Species	Station										
	g	h	i	j	k	l	m	n			
<i>Lepisosteus osseus</i>	—	—	—	—	1.22	0.97	—	—	—	—	—
<i>Dorosoma cepedianum</i>	—	—	—	—	—	6.82	—	—	—	—	—
<i>Campostoma ornatum</i>	—	—	1.16	4.85	—	—	21.46	—	—	—	—
<i>Codoma ornata</i>	2.82	—	—	21.36	—	—	—	—	—	—	—
<i>Cyprinella lutrensis</i>	—	0.65	1.16	—	4.88	16.18	35.62	29.55	—	—	—
<i>Cyprinus carpio</i>	—	—	—	—	—	2.34	—	—	—	—	—
<i>Dionda episcopa</i>	6.76	—	—	19.42	—	—	—	—	—	—	—
<i>Gila pulchra</i>	—	—	—	—	—	—	—	—	8.15	9.85	—
<i>Macrhybopsis aestivalis</i>	—	—	—	—	6.10	10.72	—	—	—	—	—
<i>Notropis braytoni</i>	—	18.95	—	—	39.02	28.46	—	—	—	—	—

Table 1. (cont.)

Species	Station											
	g	h	i	j	k	l	m	n				
<i>Notropis chiuihua</i>	1.13	21.57	10.47	4.85	—	—	—	—	—	—	—	—
<i>Notropis jemezianus</i>	—	18.97	—	—	—	11.89	—	—	—	—	—	—
<i>Pimephales promelas</i>	5.07	—	1.16	4.85	8.54	1.17	0.43	—	—	—	—	18.94
<i>Rhinichthys cataractae</i>	—	—	—	—	1.22	0.19	—	—	—	—	—	—
<i>Carpionodes carpio</i>	—	3.27	—	—	2.44	2.14	0.86	—	—	—	—	—
<i>Catostomus conchos</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Ictiobus bubalus</i>	—	—	—	—	—	0.19	—	—	—	—	—	—
<i>Cyprinotus elongatus</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Pantosteus jordani</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Pantosteus plebius</i>	—	—	—	—	—	—	—	—	—	—	—	29.55
<i>Scartomyzon austrinum</i>	—	—	—	—	1.22	—	—	—	—	—	—	—
<i>Astyanax mexicanus</i>	2.82	15.68	—	0.97	35.37	4.68	—	—	—	—	—	—
<i>Ameiurus melas</i>	—	—	—	—	—	1.36	—	—	—	—	—	—
<i>Ictalurus furcatus</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Ictalurus lupus</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Ictalurus punctatus</i>	—	—	1.16	—	—	—	—	—	—	—	—	—
<i>Pyloodictis olivaris</i>	—	—	—	—	—	0.97	—	—	—	—	—	12.12
<i>Cyprinodon eximius</i>	33.24	—	—	—	—	—	—	—	—	—	—	—
<i>Cyprinodon macroolepis</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Cyprinodon sp.1</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Cyprinodon sp.2</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Fundulus zebrinus</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Gambusia alvarezi</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Gambusia hurtadoi</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Gambusia senilis</i>	47.89	15.03	52.33	42.72	—	—	—	—	—	—	—	—
<i>Gambusia sp.1</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Gambusia sp.2</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Menidia beryllina</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Lepomis cyanellus</i>	—	—	—	—	—	1.75	—	—	—	—	—	—
<i>Lepomis gulosus</i>	—	—	—	—	—	—	—	—	—	—	—	—
<i>Lepomis macrochirus</i>	—	0.65	3.49	—	—	—	—	—	—	—	—	—
<i>Lepomis megalotis</i>	—	0.65	3.49	—	—	0.19	—	—	—	—	—	—
<i>Micropterus salmoides</i>	0.28	4.57	25.58	—	—	—	—	—	—	—	—	—
<i>Etheostoma australe/pottisi</i>	—	—	—	0.97	—	—	—	—	—	—	—	—
<i>Aplodinotus grunniens</i>	—	—	—	—	—	0.39	—	—	—	—	—	—
<i>Oreochromis aureus</i>	—	—	—	—	—	—	—	—	—	—	—	—
Specimens captured	355	116	111	103	82	513	233	132				
Number of species	8	10	9	8	9	18	7	5				
Percent introduced	0.0	0.0	0.0	0.0	4.9	2.3	0.0	0.0				

Gambusia senilis populations but with relatively few additional species, a situation that is typical of head-water creeks or degraded and polluted conditions. The species diversity was higher at the Río San Pedro site than the other two localities; however, all were small streams dominated by *G. senilis*.

The lower reaches of Río Conchos tributaries (Stations 4, 5, 7, and 8a) had relatively large numbers of *N. jemezianus*, *G. senilis*, and usually *Camptostoma ornatum*. These sites differed from each other in that the Río Santa Isabella localities also contained large numbers of *Codoma ornata* and our only captures of *Gila pulchra*, whereas the two sites near San Diego de Alcalá had relatively large numbers of *Dionda episcopa* and either *C. eximius* (Station 8a) or an undescribed species of *Cyprinodon* (Station 7). The hot spring outflows at the San Diego de Alcalá location (Station 7) contained our largest collection of *Lepomis macrochirus*, an introduced species accounting for 18% of the fish collected at that site.

The Río Conchos immediately above Presa de Boquilla (Station 2) had relatively large numbers of *C. lutrensis*, *N. chihuahua*, *C. ornatum* and *G. senilis* and this area also contained *P. promelas*, *A. mexicanus*, *L. cyanellus*, and the introduced *Oreochromis aureus*. Although this area has been impacted greatly by urbanization and influences from the reservoir, introduced species were less abundant than at some of the other localities.

The downstream and middle Río Conchos stations (6 and 10) were heavily impacted by channelization and degraded water quality from agricultural inputs. The collections were dominated by *C. lutrensis*, which accounted for a third to about 60% of the total fish captures at these sites. Also present were the introduced *Menidia beryllina* and some of the more typical Chihuahuan Desert fishes, such as *N. chihuahua*, *N. braytoni*, *Scartomyzon austrinus* and several catfish, including *Ictalurus furcatus*, *I. lupus*, *I. punctatus*, and *Pylodictis olivaris*. These sites also produced large numbers of the introduced *Oreochromis aureus*, especially the site near Julimes (Station 6), which was dominated by this species.

COMPARISON WITH 1954 DATA

To contrast changes in fish community composition that have occurred in the half-century since the 1954 collections were taken, we summarized collections from Río Conchos stations, Río San Pedro stations and Río Florido stations taken during the 1954 series of collections and our present samples. The high degree of overall similarity in species occurrences indicates that the fish fauna in the Río Conchos basin is still intact (Table 2). Some notable changes have occurred. Introduced fishes have long been known from the Río Conchos. For example, the common carp (*Cyprinus carpio*) was present in the early collections. However, blue tilapia (*Oreochromis aureus*) were not in the system in the 1950s, but are now widely found throughout and dominate the fish assemblages at some localities. There appears to be a change in other elements of the fish assemblages, as well, possibly in response to lessened water flows in the basin. There appears to be a loss of minnow diversity (10 species commonly found in 1954 versus 5 in the present study) and a diminution of species commonly

found in large river systems (*Lepisosteus osseus*, *Notropis braytoni*, *N. jemezianus*, *Cycleptus elongatus*, *Aplodinotus grunniens*, *Micropterus salmoides*, *Lepomis macrochirus*, and *L. megalotis*). In contrast, there appear to have been increases in smaller stream forms such as *Cyprinodon eximius* and *Gambusia senilis* both of which are indicative of diminishing and regulated flows. As these latter two species are of conservation concern, their increased populations in our present collections could be tenuous. Further declines in streamflow could negatively impact these species. The change in the fish communities of the Río Florido is dramatic. The fish assemblage in this stream was quite similar to the mainstem Río Conchos in 1954, but in our survey the river was dry.

Our two samples from the Río Chuisca (Stations 8a and 8b) in 1994 and 1995, indicate how swiftly noticeable changes can occur in fish assemblages. During our initial sampling, workers were laying a water pipe as part of a system to pump water out of the

Table 2. Comparison of changes in fish communities between 1954 and 1994-1995 for selected streams in the Rio Conchos basin. An asterisk (*) indicates that the stream was dry.

Species	Río Conchos		Río San Pedro		Río Florido	
	1954	1994-95	1954	1994-95	1954	1994-95
<i>Lepisosteus osseus</i>	0.60	—	—	—	—	*
<i>Dorosoma cepedianum</i>	1.02	—	—	—	—	*
<i>Campostoma ornatum</i>	—	1.04	0.39	14.42	8.97	*
<i>Codoma ornata</i>	—	0.18	1.46	20.29	8.64	*
<i>Cyprinella lutrensis</i>	9.47	3.31	0.39	—	22.69	*
<i>Cyprinus carpio</i>	0.81	—	—	—	—	*
<i>Dionda episcopa</i>	6.21	7.23	3.51	—	13.58	*
<i>Gila pulchra</i>	—	—	—	6.01	3.30	*
<i>Macrhybopsis aestivalis</i>	1.78	—	—	—	—	*
<i>Notropis braytoni</i>	9.10	—	—	—	—	*
<i>Notropis chihuahua</i>	8.79	14.24	4.07	40.99	5.99	*
<i>Notropis jemezianus</i>	15.21	—	—	0.40	0.35	*
<i>Pimephales promelas</i>	9.77	0.26	3.02	2.40	8.38	*
<i>Rhinichthys cataractae</i>	0.12	—	—	—	2.21	*
<i>Carpionodes carpio</i>	1.12	—	—	—	2.40	*
<i>Catostomus conchos</i>	—	0.04	—	—	2.84	*
<i>Cycleptus elongatus</i>	0.24	—	—	—	—	*
<i>Ictiobus bubalus</i>	—	0.02	—	—	—	*
<i>Scartomyzon austrinus</i>	0.43	0.04	—	—	—	*
<i>Astyanax mexicanus</i>	7.74	3.90	1.46	—	4.97	*
<i>Ameiurus melas</i>	—	—	—	0.13	—	*
<i>Ictalurus furcatus</i>	0.28	—	—	—	—	*
<i>Ictalurus lupus</i>	—	0.02	—	—	—	*
<i>Ictalurus punctatus</i>	—	—	0.39	—	—	*
<i>Pylodictis olivaris</i>	0.40	—	—	—	—	*
<i>Cyprinodon eximius</i>	1.67	15.46	17.25	—	0.18	*
<i>Gambusia speciosa</i>	1.42	—	—	—	—	*
<i>Gambusia senilis</i>	4.19	26.97	57.06	10.95	13.62	*
<i>Gambusia sp.2</i>	—	23.96	—	—	—	*
<i>Lepomis cyanellus</i>	0.32	0.02	—	—	—	*
<i>Lepomis macrochirus</i>	9.89	0.18	1.16	—	—	*
<i>Lepomis megalotis</i>	2.58	0.02	1.16	2.00	1.88	*
<i>Micropterus salmoides</i>	6.57	2.22	8.67	—	—	*
<i>Etheostoma australe/pottsi</i>	0.23	—	—	2.40	—	*
<i>Aplodinotus grunniens</i>	0.04	—	—	—	—	*
<i>Oreochromis aureus</i>	—	0.89	—	—	—	*
Number of collections	13	3	3	1	5	*
Number of species	26	19	13	10	15	0

stream. A year later, the pump had been installed and the stream was visibly altered, showing numerous effects of the construction activity. Relative abundances of *C. ornatum*, *D. episcopa*, *N. chihuahua*, and *C. eximius* were lower and those for *C. lutrensis* and *G. senilis* were notably higher during the second visit. *Pimephales promelas*, *A. mexicanus*, *L. cyanellus* and

O. aureus also had somewhat increased relative abundances following the perturbation. The species showing increased relative abundances are characteristic of either highly degraded areas or small headwater streams and generally are considered to be more tolerant of extreme environmental conditions.

STATUS OF SPECIES

A number of fishes inhabiting the Chihuahuan Desert region have been proposed for listing as endangered or threatened species of the U.S. or Mexico. Based on our results we provide additional observations on the status of some of these species.

The Mexican stoneroller (*Campostoma ornatum*) occurs in numerous localities in the Big Bend region of Texas and northwestern Mexico, including the ríos Conchos, del Fuerte, Casas Grandes, del Carmen, Yaqui, Papigóchic, Sonora, Nazas, Piaxtla and Trujillo (Burr, 1976). Although it occurs throughout the Río Conchos basin, we only found it abundant in the Río Santa Isabella. Some populations are seemingly ephemeral, particularly in highly impacted habitats such as Río Chuviscar. In 1994, this species had a relative abundance of 2.4%, but in 1995 no specimens were obtained. Contreras-B. (1977) reported it extirpated from the Río Chihuahua (= Chuviscar) and the Río Conchos at Camargo, citing the loss of well-oxygenated, clear, moving water flowing over sand and gravel bottoms due to lowered water tables, siltation and sewage effluent. Our Río Conchos sample at Julimes was downstream of Camargo and we did not obtain *C. ornatum*. However, our Río Conchos sample at Valle de Zaragoza is upstream of Camargo and there we obtained 31 specimens. Our collections support a threatened status for this species and agree with many of the designations and reasons for this status given by various governmental agencies and researchers (Miller, 1972; Williams et al., 1989; Hubbs et al., 1991; Texas Organization for Endangered Species, 1995; CONABIO, 1997).

The Chihuahua shiner (*Notropis chihuahua*) is listed as Threatened by the Texas Parks and Wildlife Department, Miller (1972), Hubbs et al. (1991) and the Texas Organization for Endangered Species (1995). Our findings agree with Burr's (1980) assessment that the species occurs sporadically in Texas in the Big Bend region of the Río Grande, but is currently abundant in tributaries of the Río Conchos. Previous findings from studies in the Big Bend region of the Río Grande range from absence of the species (Platania, 1990; IBWC, 1994) to a relative abundance of less than approximately 1% (Hubbs and Wauer, 1973; Hubbs et al., 1977; Bestgen and Platania, 1988).

The Río Grande shiner (*Notropis jemezianus*) is listed as Rare by Mexico (CONABIO, 1997), Threatened by the Texas Organization for Endangered Species (1995), Threatened by Hubbs et al. (1991) and Special Concern by Williams et al. (1989). The historic range included the Río Grande, Pecos River (New Mexico and Texas), and ríos Conchos, San Juan and Salado drainages of Mexico (Gilbert, 1980). Hubbs (1940) noted that the species was "characteristic of the Río Grande and its tributaries in New Mexico, Texas and northeastern Mexico" and Treviño-Robinson (1959) found the species throughout the middle Río Grande of the Texas-Mexico borderlands, almost to the mouth of the Río Grande during her studies in the 1950s. However, the range of *N. jemezianus* in the Río Grande and Pecos River has declined dramatically, (Edwards and Contreras-B., 1991; Hubbs et al., 1991; Edwards et al., 2001). This species is part of a main-stream Río Grande-Río Conchos faunal assemblage that is not dependent on tributaries (Hubbs et al., 1977). It is typically found in large, open rivers over sand and gravel (Gilbert, 1980) where current flows keep the substrate clean from accumulated silt. Our collections support the Threatened status designation given this species by Hubbs et al. (1991).

The headwater catfish (*Ictalurus lupus*) is listed as Rare by Mexico (CONABIO, 1997), Watch List by the Texas Organization for Endangered Species (1995), and Special Concern by Williams et al. (1989) and Hubbs et al. (1991). In our study we found this little known species only in the Río San Pedro and the Río Conchos at Cuchillo Parado, Julimes and Zaragoza, where it was always in low abundance. These results support the Rare (and Watch List) status designations previously given to this species.

Hubbs et al. (1991) listed the undescribed Chihuahuan catfish (*Ictalurus* sp.) as Special Concern. Very little is known about this very cryptic and rare species and none were obtained in our collections. It occurred historically in the Río Grande basin of New Mexico and Texas, the Río Conchos basin in Chihuahua and the Río San Fernando in Tamaulipas. The absence of this species in our samples may indicate that the species is in greater danger of extinction than generally understood.

The Conchos pupfish (*Cyprinodon eximius*) is listed as Threatened by the Texas Parks and Wildlife Department, the Texas Organization for Endangered Species (1995), Williams et al. (1989), Hubbs et al. (1991), and Mexico (CONABIO, 1997). Historically this species was widely distributed, occurring in the upper Río Conchos and Río Sauz in Chihuahua and Alamito, Terlingua and Tornillo creeks and Devils River in Texas (Miller, 1976, 1981; Hubbs and Echelle, 1973; Minckley, 1980; Hubbs et al., 1991). The population in Dolan Creek, a tributary of the Devils River, was extirpated in 1958 and successfully reestablished in 1979 (Garrett, 1980; Hubbs and Garrett, 1990; Garrett et al., 1992). In our surveys, the species was abundant in the Río Chuviscar and occurred in low numbers in headwater streams and tributaries of the Río Conchos. The population in the Río Sauz Basin may have been extirpated (Echelle et al., this volume).

The blotched gambusia (*Gambusia senilis*) is listed as Threatened by Miller (1972), Mexico (CONABIO, 1997), and Texas Parks and Wildlife Department, Spe-

cial Concern by Williams et al. (1989) and Extirpated in Texas by Hubbs et al. (1991) and the Texas Organization for Endangered Species (1995). The historic range of the blotched gambusia includes the Río Conchos Basin and Devils River (Hubbs, 1958; Guillory, 1980). Although Hubbs and Springer (1957) reported its range as the Río Conchos downstream as far as Julimes, our collections at Julimes contained no *G. senilis*. However, an abundant population was present farther downstream in the Río Chuviscar and the species almost completely dominates the fish community in the headwaters northwest of Ciudad Chihuahua. In general, we found *G. senilis* abundant and widely distributed in our Mexican samples. The Texas population was isolated by Amistad Reservoir and ultimately eliminated (Hubbs and Echelle, 1973; Hubbs et al., 1991). The Rio Grande Fishes Recovery Team has recommended reestablishment of the Texas population in Devils River State Natural Area from stocks in the Río Chuviscar. A Threatened status seems appropriate for the existing populations of this species.

DISCUSSION

Desert ecosystems are easily perturbed and often slow to recover. Entrenchment of streams from erosion due to overgrazing and deforestation (Ohmart and Anderson, 1982), introductions of exotic species, and extinction of native species may all cause permanent damage to these systems. While other perturbations such as pollution, reduced groundwater, and dam construction are theoretically recoverable, the return to a pristine state is unlikely.

Anthropogenic changes in the Río Conchos basin have been going on since the mid-1800s (Miller, 1961, 1977) but the effects have been compounded over time and are now becoming dramatic. Our survey indicates detrimental impacts on the fish assemblages of the Río Conchos in the past 40 years. In the early part of the 20th century it was apparent that water was becoming a major problem in Chihuahua as extensive irrigation projects were initiated (Tamayo and West, 1964). Brand (1937) noted for northwestern Chihuahua that "the increasing use of spring and river water for irrigation on the haciendas and colonias of the region has contributed markedly to the lessened

flow of the rivers in their lower courses." At least 30 springs have gone dry in Chihuahua and Coahuila and river discharges of the Río Nazas, Bolson Mayrán, Río Aguanaval, Bolson Viesca, Río de Nadadores, Río Saltillo, Río Salinas, Río del Carmen and the middle Rio Grande are reduced (Contreras-B. and Lozano-V., 1994). Under these conditions of decreased flow, droughts are even more devastating because of increased groundwater pumping for agricultural and municipal use. Such extreme conditions favor more tolerant species often at the expense of less widespread species. Tributary creeks tend to be impacted more severely, yet these areas are critical to the breeding and rearing of young of many of the indigenous species including *C. ornatum* and *N. chihuahua* in the Rio Grande (Hubbs and Wauer, 1973).

One factor noticed during our collections in the Río Conchos Basin was the great efforts toward modernization of the infrastructure in Chihuahua including its highways and municipal facilities. Few areas have escaped this leap toward modernization. However, the magnitude of change for the natural aquatic sys-

tems is sometimes quite dramatic. Many springs, especially those located near human habitation, have been modified or are being modified into swimming areas or spas. In the larger municipalities there is a reasonably strong message of water conservation, but the surrounding countryside shows many signs of increasing use of water consumptive measures such as flood irrigation and spray-water delivery systems.

The Río Conchos Basin has been impacted by centuries of human habitation. Exploitation of limited resources, particularly groundwater pumping, has degraded that environment, caused extirpation and extinction of species and, ultimately, loss of habitat and ecosystems (Smith and Miller, 1985). We suspect that the fish assemblages of this region are indicators of the overall integrity of the ecosystem. The few remaining relatively pristine localities need careful management if they are to be preserved.

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HISTORICAL AND RECENT FISH FAUNA OF THE LOWER PECOS RIVER

Christopher W. Hoagstrom

ABSTRACT

The lower Pecos River extends 770 km, crossing the Permian Basin and Edwards Plateau from 17 km northwest of Carlsbad, New Mexico to the Rio Grande, near Langtry, Texas. Recent (1991 to 1999) fish collections were depauperate compared with historical collections from the area. Recent composition of the ichthyofauna was divisible into three assemblages, associated with the following river segments: between Brantley and Red Bluff dams, Red Bluff Dam to Live Oak Creek confluence, and Live Oak Creek to the Rio Grande, but this segregation was not evident within historical collections. Historically, 27 native fish species occurred in all three segments, whereas in recent collections only nine native fishes occurred in all three. Recent native fish species richness was reduced between 47 and 54% (by segment) from historical collections. Riverine species were poorly represented in recent collections. Seven native and one nonnative fishes historically represented the genus

Notropis, but only three species of the genus were found in recent collections. Incidental stockings and bait bucket releases established nonnative euryhaline fishes (*Fundulus grandis*, *Menidia beryllina*) that represented significant proportions of recent collections. A similar introduction resulted in replacement of *Cyprinodon pecosensis* by a hybrid swarm (*C. pecosensis* x *C. variegatus*). Nonnative game fishes represented a minor portion of recent collections, despite concerted efforts to establish several species. Diminishing springflows, nonnative fish introduction and spread, and toxic algal blooms further threaten native fish populations, while habitat and water quality deterioration favor nonnative species to the detriment of natives. Appreciation of lower Pecos River historical significance to native Rio Grande fishes is important for promoting conservation of remnant native species assemblages both within the lower Pecos River and throughout the Rio Grande basin.

INTRODUCTION

The degraded condition of the lower Pecos River in recent decades was evidenced by elevated salinity (Davis, 1987), toxic algal blooms (James and De la Cruz, 1989; Hubbs, 1990; Rhodes and Hubbs, 1992), and replacement of a native pupfish by a hybrid swarm with a nonnative congener (Echelle and Connor, 1989). Nineteenth century accounts of the lower Pecos River (Pope, 1854; Dearen, 1996) are very different from

more recent accounts (Grozier et al., 1966; Davis, 1980), suggesting that the historical fish fauna may have also been different from the present fauna. This paper summarizes historical fish records, compares them with recent (1991 to 1999) records, and provides a historical perspective on habitat and faunal conditions recently observed.

STUDY AREA

The Pecos River is the largest Rio Grande tributary in the United States. The mainstem extends 1,490 km, from more than 3,960 m above sea level in the Sangre de Cristo Mountains, New Mexico, to roughly 305 m above sea level at the river mouth on the Texas-

Coahuila border (Figure 1). In the Pecos River drainage, cold- and cool-water streams are present in montane headwaters of the mainstem and tributaries. The middle and lower Pecos basins (Figure 1) include warm-water habitats such as the Pecos River

Lower Pecos River

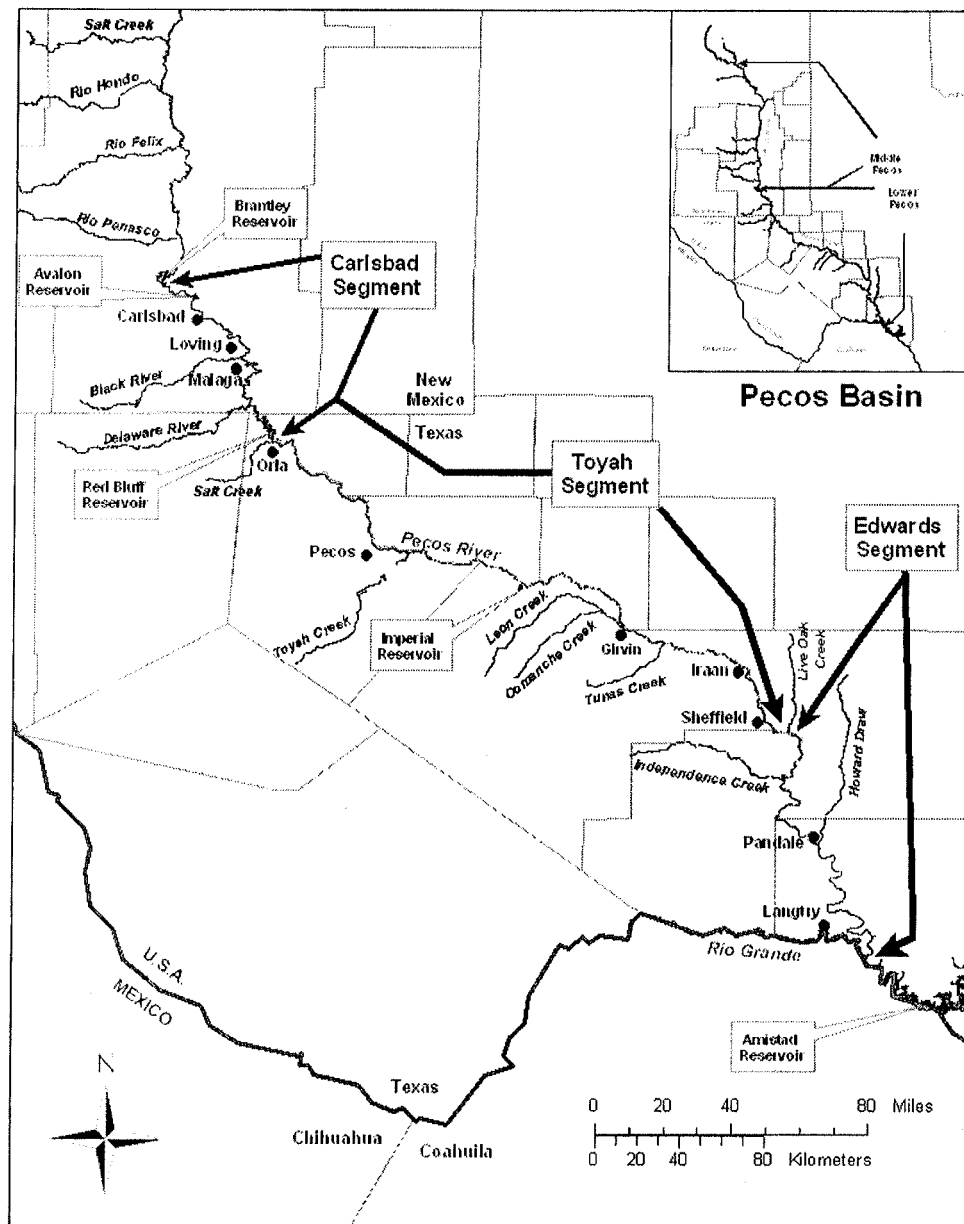


Figure 1. Map of Lower Pecos River, New Mexico and Texas, with the three fish faunal segments delineated. Also included are selected reservoirs, tributaries, and cities.

mainstem, spring-fed tributaries, and spring-fed/flood-plain wetlands. This paper is solely concerned with the lower Pecos River mainstem.

The lower Pecos River extends roughly 770 km, crossing the Permian Basin for about 575 km (Hill, 1996) and then the Edwards Plateau for 195 km (King, 1935). The Pecos River, Permian Basin section begins 17 km northwest of Carlsbad, New Mexico

(Kelley, 1971; Bachman, 1980), where the river traverses Barrera del Guadalupe, extending downstream to the Edwards Plateau, near Iraan, Texas (Anaya, 2001). Within this stretch, the Pecos River crosses a series of alluvial basins (Maley and Huffington, 1953; Jones, 2001). Downstream, the Pecos River is incised within the Edwards Plateau and confined by limestone cliffs (Thomas, 1972).

The lower Pecos River receives surface inflow from the middle Pecos River (Figure 1) and from local tributaries that originate in mountains to the west (Guadalupe, Delaware, Apache, Davis, Barrilla, Del Norte, and Glass mountains) (Hill, 1996). However, even the largest western tributaries (e.g., Dark Canyon, Black River, Delaware River, Salt Creek, Toyah Creek, Coyanosa Draw, Tunis Creek) sustain only intermittent surface flow. Headwater flows normally sink into the ground at the base of the mountains, but rise to the surface downstream where water-bearing strata outcrop or encounter impermeable strata (Brune, 1981). A number of significant aquifers, present throughout the lower Pecos River basin, interact hydrologically (discharge or recharge) with surface waters (Richey et al., 1985; Hill, 1996; Mace et al., 2001). An exceptional example is the regional flow system that extends west of the surface drainage boundary and apparently distributes groundwater to multiple Pecos River tributaries by interconnection of the following aquifer basins: Ryan Flat, Lobo Flat, Salt Basin, Apache Mountains, Balmorhea Basin, and Toyah Basin (Sharp, 2001). The east edge of the lower Pecos

River drainage is bounded by the Southern High Plains from which no major tributaries enter (Lee, 1925), because of high percolation rates into surficial sands (Jones, 2001). Within the Edwards Plateau, spring-fed tributaries (e.g., Live Oak Creek, Independence Creek, Howard Creek) join the river from both east and west (Brune, 1981). The lower Pecos River terminates at the Rio Grande, near Langtry, Texas (Figure 1).

The lower Pecos River forms the boundary between the Kansan, Balconian, and Chihuahuan biotic provinces, while the Navahonian province delineates the northwestern boundary and the Tamaulipan province extends up the Rio Grande to near the Pecos River confluence (Blair, 1950). Fishes representative of all five provinces occupy the lower Pecos River, accounting for a relatively diverse native fish fauna (Hubbs, 1957; Smith and Miller, 1986). Most of the lower Pecos River Basin lies within the Chihuahuan province where many native species are threatened or endangered (Edwards et al., 1989).

HISTORY

According to historical accounts, the lower Pecos River was deep and swift, with steep, unstable banks, a shifting sand substrate, and abundant quicksand (Pope, 1854; U.S. Geological Survey, 1900; Brune, 1981; Leftwich, 1987; Dearen, 1996). In 1854, Pope reported, "The Pecos traverses its valley in a very tortuous course, and with a current of about two and a half miles to the hour, and from five to twenty feet depth of water." Rapids or falls were present wherever the Pecos River encountered bedrock or where tributaries discharged gravel and boulders (Pope, 1854; Hufstetler and Johnson, 1993; Dearen, 1996). Pecos River water was often turbid and had relatively high mineral content, giving the river a reputation for having only bad water. Some springs in the area were very salty or sulphurous (Pope, 1854; Brune, 1981) and saline wetlands were common within the floodplain (Schroeder and Matson, 1965; Leftwich, 1987), but Pope (1854) noted "Although the water of the Pecos is somewhat salty. . . the use of it has not been followed by any injurious consequences to the health, of a serious character."

Anglo-American settlement along the lower Pecos River began in the 1860s and intensified with the coming of railroad and irrigation companies (Lingle and Linford, 1961). Large-scale water development began at the head of the lower Pecos River with Avalon (1891) and McMillan (1893) dams (U.S. Geological Survey, 1900; Freeman and Bolster, 1911; Grover et al., 1922; Meinzer et al., 1926; U.S. National Resources Planning Board, 1942). Immediately thereafter, numerous river diversions were established between Carlsbad, New Mexico, and Girvin, Texas (Taylor, 1902; Grover et al., 1922; U.S. National Resources Planning Board, 1942). Sediment deprivation resulting from McMillan and Avalon dams changed Pecos River substrate from sand to bedrock (in swift areas) and silt (in slow areas). With capture and diversion of surface flows, groundwater springs became the primary source of flow in the mainstem lower Pecos River (Taylor, 1902; Grover et al., 1922; Robinson and Lang, 1938). For example, in May 1918, the Pecos River gained 47.8 cubic feet per second (cfs) (1.35 cms) between the New Mexico/Texas border and

Girvin, Texas (Grover et al., 1922). In 1925, W.T. Lee observed “[the Pecos River] is a stream of considerable size at all times. . . Records of two gauging stations near Carlsbad, 2.5 miles apart, show that 80 second-feet of water enters the river in this distance.” Once the lower Pecos River was fragmented, each river segment developed water quality and flow characteristics specific to local conditions (i.e., the continuity of the mainstem river environment was disrupted) (U.S. National Resources Planning Board, 1942).

Additional mainstem reservoirs were established during a second development period, beginning in the 1930s (Lingle and Linford, 1961). Red Bluff Dam (1936), near the New Mexico/ Texas border, supplied seven irrigation districts in Texas and Sumner Dam (1937) of the middle Pecos River supplied Carlsbad Irrigation District. Both facilities captured floodwaters and sediments. Even with new storage facilities, expanding development and drought caused irrigators to increasingly rely on groundwater, particularly after 1942 (Thomas et al., 1963; West and Broadhurst, 1975). As a result, groundwater flow within adjoining aquifers was altered (Thomas et al., 1963; Mace et al., 2001). Particularly heavy groundwater pumping in the Permian Basin altered groundwater flow-paths, virtually eliminating historical base-flow gains in most areas and causing significant base flow losses in some (Grozier et al., 1966; West and Broadhurst, 1975; Hiss, 1980; Brune, 1981).

Davis (1987) summarized a dramatic, human-induced increase in total dissolved solids (i.e., salinity) between 1938 and 1981. Flood control (Howard, 1942; Davis, 1980, 1987), stratification in impoundments and riverine pools (Davis, 1980, 1987), saline aquifer intrusion (Hood, 1963; Havenor, 1968; Jones, 2001), irrigation return flows (Robinson and Lang, 1938; U.S. National Resources Planning Board, 1942; LaFave, 1987; Ashworth, 1990; Mace et al., 2001), oil field pollution (Wiebe et al., 1934; Campbell, 1959; Grozier et al., 1966; Ashworth, 1990), and *Tamarix* (Davis, 1987), each contributed to salinity increase. Because each of these factors was initiated prior to 1935, when the first water quality investigations were conducted (Robinson and Lang, 1938; U.S. National Resources Planning Board, 1942), pre-development salinity is unknown, but was presumably lower than first recorded in 1935.

For purposes of this paper, the lower Pecos River was divided into three segments (Figure 1): 1) Carlsbad segment: McMillan Dam (replaced by Brantley Dam in 1988) to Red Bluff Dam; 2) Toyah segment: Red Bluff Dam to Live Oak Creek confluence; and 3) Edwards segment: Live Oak Creek confluence to the Rio Grande. Segments were distinguishable from each other with respect to flow regime, geomorphology, and water chemistry (Hillis, unpublished; Davis, 1980, 1987; Sublette et al., 1990; Rhodes and Hubbs, 1992).

HISTORICAL FISH SURVEYS

Ichthyological surveys and summaries seldom have considered the lower Pecos River as a unit. Miller (1977) provided the only species list for fishes specific to the area. Important summaries included Evermann and Kendall (1894), who listed fishes taken from the Pecos River basin during railroad and boundary surveys, and Smith and Miller (1986), who listed fishes native to the entire Pecos River basin and discussed their zoogeographic origins. This paper primarily follows Smith and Miller (1986) in designation of fish species as native (but see Table 1).

Historical lower Pecos River fish surveys were not equal among segments. The Carlsbad segment was most heavily surveyed and included the earliest

lower Pecos River collections (Pope [1854] at Delaware River confluence [Evermann and Kendall, 1894]). Extensive New Mexico Department of Game and Fish (NMDGF) surveys between 1955 and 1970 captured fish using a fish barrier trap and gill-nets below McMillan Dam (Navarre, 1959, 1960; Little, 1961b, 1963a, 1963b). Gill-nets were also used on the mainstem river between Tansill Dam in the city of Carlsbad and Red Bluff Reservoir (Little, 1964c, 1964d, 1965), and within Avalon Reservoir, Carlsbad Municipal Lake, and Red Bluff Reservoir (Little, 1960a, 1960b, 1961a, 1964b, 1964c). Fishes were also salvaged from irrigation canals (Little, 1964a). General fish community surveys, using seines, were conducted by Koster and associates, University of New Mexico (Koster,

Table 1. Native fishes known from the mainstem lower Pecos River. Inclusion of species as native follows Smith and Miller (1986) except where noted (numbered footnotes). Recent status of native fishes, based on 1991 to 1999 collections is given: Thriving = frequent and widespread collections in high number (thousands); Stable = reproductive populations in moderate numbers (hundreds); diminished = range reduced, occurrence in moderate numbers; Tenuous = few collections, low numbers (<25); ? = absent from collections but possible via dispersal from Rio Grande; ?? = undetermined due to difficulty in identification and/or lack of documentation; Absent = missing from recent collections, with the year of most recent collection from the lower Pecos River given for each species. The known historical and recent distribution of each species is given by segment. Lettered footnotes provide references of taxonomic interest. Names and taxonomic order follow Mayden et al. (1992).

Species	Recent Status	Historical Distribution	Recent Segment Distribution
<i>Atractosteus spatula</i> ¹	ABSENT - 1958	EDWARDS?	Unknown, possible in Edwards
<i>Lepisosteus oculatus</i> ²	ABSENT - 1958	TOYAH - EDWARDS?	Unknown, possible in Edwards
<i>Lepisosteus osseus</i>	STABLE	ALL	All, abundant in Carlsbad
<i>Anguilla rostrata</i>	ABSENT - 1948	ALL	-
<i>Dorosoma cepedianum</i>	THRIVING	ALL	All, abundant in Carlsbad & Toyah
<i>Campostoma anomalum</i> [*]	ABSENT - 1958	EDWARDS	-
<i>Cyprinella lutrensis</i>	STABLE	ALL	All, abundant in Carlsbad
<i>Cyprinella proserpina</i>	STABLE	TOYAH - EDWARDS	Edwards, common
<i>Dionda episcopa</i>	DIMINISHED	ALL	Edwards, associated with springs
<i>Hybognathus amarus</i>	ABSENT - 1963	CARLSBAD - TOYAH	-
<i>Macrhybopsis a. aestivalis</i> ^a	TENUOUS	ALL	Toyah & Edwards, rare
<i>Notropis amabilis</i> ^{ba}	TENUOUS	ALL	Edwards, uncommon
<i>Notropis braytoni</i>	TENUOUS	ALL	Toyah & Edwards, rare
<i>Notropis buechanani</i>	ABSENT - 1965	EDWARDS	-
<i>Notropis jemezianus</i>	ABSENT - 1987	ALL	-
<i>Notropis l. ludibundus</i> ^c	TENUOUS	CARLSBAD - TOYAH	Toyah, rare
<i>Notropis orca</i> ^d	ABSENT - 1940	EDWARDS	-
<i>Notropis simus pecosensis</i> ^d	ABSENT - 1987	CARLSBAD	-
<i>Phenacobius mirabilis</i> ^e	??	?	-
<i>Pimephales promelas</i>	STABLE	ALL	Carlsbad, abundant
<i>Pimephales vigilax</i>	STABLE	TOYAH - EDWARDS	Edwards, common
<i>Rhinichthys cataractae</i> [*]	ABSENT - 1947	CARLSBAD	-
<i>Carpionodes carpio elongatus</i> ^e	DIMINISHED	ALL	Carlsbad, common
<i>Cycleptus elongatus</i> ^f	DIMINISHED	CARLSBAD, EDWARDS	Carlsbad, mostly between Brantley & Avalon dams, possible in Edwards
<i>Ictiobus bubalus</i>	TENUOUS	ALL	Carlsbad, rare, possible in Edwards
<i>Ictiobus niger</i> ^g	??	?	Unknown, confusing taxonomy
<i>Scartomyzon congestus</i>	DIMINISHED	ALL	Carlsbad & Edwards, common in Carlsbad
<i>Astyanax mexicanus</i>	DIMINISHED	ALL	Carlsbad & Edwards, uncommon
<i>Ictalurus furcatus</i> ^g	ABSENT - 1958	ALL	Unknown, possible in Edwards
<i>Ictalurus lupus</i>	??	ALL	Unknown, difficult identification and taxonomy
<i>Ictalurus punctatus</i> [^]	DIMINISHED	ALL	All, most common in Carlsbad
<i>Pylodictis olivaris</i> [^]	TENUOUS	ALL	Carlsbad & Toyah, rare
<i>Fundulus zebrinus</i>	ABSENT - 1993	ALL	-
<i>Lucania parva</i> ^h	THRIVING	ALL	All, most abundant in Toyah
<i>Cyprinodon pecosensis</i>	ABSENT - 1994	ALL	-
<i>Gambusia affinis</i> ⁷	THRIVING	ALL	All, abundant in Carlsbad & Toyah
<i>Gambusia speciosa</i>	STABLE	EDWARDS	Edwards
<i>Lepomis cyanellus</i>	DIMINISHED	ALL	All, common in Carlsbad
<i>Lepomis gulosus</i>	DIMINISHED	ALL	Carlsbad
<i>Lepomis macrochirus</i> [^]	STABLE	ALL	Carlsbad & Toyah, common in Carlsbad
<i>Lepomis megalotis</i>	DIMINISHED	ALL	All, rare in Toyah
<i>Micropterus salmoides</i> [^]	DIMINISHED	ALL	All, uncommon
<i>Etheostoma grahami</i>	TENUOUS	EDWARDS	Edwards, uncommon
<i>Etheostoma lepidum</i>	ABSENT - 1992	CARLSBAD	-

Table 1. (cont.)

Species	Recent Status	Historical Distribution	Recent Segment Distribution
<i>Percina macrolepida</i>	TENUOUS	CARLSBAD - TOYAH	Carlsbad, between Brantley & Avalon dams, Black River confluence
<i>Aplodinotus grunniens</i>	TENUOUS	TOYAH - EDWARDS	Toyah, rare, possible in Edwards
<i>Cichlasoma cyanoguttatum</i>	TENUOUS	TOYAH - EDWARDS	Toyah & Edwards, rare

¹Not included by Smith and Miller (1986), but known from at least Amistad Reservoir (J.F. Scudday, Sul Ross State University, pers. comm.), possibly upstream to New Mexico (Hubbs, 1957).

²Not included by Smith and Miller (1986), but well known from the lower Pecos River throughout Texas (e.g., Campbell, 1959; Hillis, unpublished).

³Incorrectly listed as extinct by Smith and Miller (1986; Hubbs et al., 1991).

⁴Not included by Smith and Miller (1986), but known from a collection by R.M. Bailey at U.S. Highway 90 bridge (Chernoff et al., 1982).

⁵Listed as native by Smith and Miller (1986), but nonnative by Sublette et al. (1990). Very few historical records (see Campbell, 1959).

⁶Listed by Koster (1957) and also by Smith and Miller (1986). The author has found no reference to extant museum specimens or credible accounts.

⁷Not included by Smith and Miller (1986), but considered native to the Pecos River (Rauchenberger, 1989 and many others).

⁸Rio Grande speckled chub, following Eisenhour (1997).

⁹Pecos River variant (low mean scale radii counts), following Coburn (1982).

¹⁰Use of nominal subspecies name follows Tanyolac (1973) as modified by Mayden and Gilbert (1989).

¹¹Pecos bluntnose shiner, following Chernoff et al. (1982).

¹²Slender carpsucker, following Hubbs and Black (1940).

¹³Rio Grande basin variant, following Burr and Mayden (1999) and Buth and Mayden (2001).

¹⁴Rio Grande basin variant (unique spotting and head shape), see Garrett and Edwards (2001).

¹⁵Pecos River race, following Hubbs and Miller (1965).

¹⁶Native to Pecos basin, but may be accidental or introduced in lower Pecos River.

¹⁷Native species that were also introduced as game-fish, likely from sources outside the lower Pecos River basin

1957; Museum of Southwestern Biology records) and, subsequently, by Sublette and associates, Eastern New Mexico University (Sublette, 1975; Hatch, 1985; Sublette et al., 1990). Comprehensive information on rare native fishes of New Mexico was provided by Hubbs and Echelle (1972), while Hatch (1985) and Sublette et al. (1990) summarized historical fish collections for that state.

Historical fish surveys were least extensive in the Toyah segment (Table 2). Collections, primarily by Bailey and others (University of Michigan Museum of Zoology records), Texas Parks and Wildlife Department (TPW; Campbell, 1959), and Hubbs and Springer, University of Texas at Austin (UT-Austin; Hubbs, 1954, 1957; Texas Natural History Collection records), provided data on fish distribution between 1940 and 1960. Later collections by Davis in 1976, Texas Department of Water Resources (Davis, 1981), Hillis in 1979, and Rhodes et al. in 1987-1988, both of

UT-Austin (Hillis, unpublished; Hillis et al., 1980; Rhodes and Hubbs, 1992; Texas Natural History Collection records), and Linam and Kleinsasser, TPW in 1987 (Linam and Kleinsasser, 1996) provided more recent information, but only Linam and Kleinsasser surveyed upstream of Girvin, Texas. Historical Toyah segment fish surveys relied entirely on seines.

Historical fish surveys from the Edwards segment were less extensive than the Carlsbad segment, but greater than from the Toyah segment (Table 2). Except for a TPW survey (Campbell, 1959), surveys were primarily conducted by UT-Austin associates (Hubbs, 1954, 1957; Treviño-Robinson, 1955, 1959; Hillis et al., 1980; Rhodes and Hubbs, 1992; Texas Natural History Collection records) and Tulane University associates (Tulane University Museum of Natural History records) (Table 3). Historical Edwards segment fish surveys relied on seines except for three gill net collections reported by Campbell (1959).

MATERIALS AND METHODS

The historical (1939-1990) fish fauna of the lower Pecos River was summarized using published literature, agency reports, museum records (Museum of Southwestern Biology, University of Michigan Museum of Zoology, Texas Natural History Collection, Tulane University Museum of Natural History), and personal communications (G.P. Garrett, TPW; J.P. Karges, The Nature Conservancy; S.P. Platania, Museum of Southwestern Biology; J.F. Scudday, Sul Ross State University). Occurrence of fishes in historical collections was tabulated for each of the three lower Pecos River segments. Recent (1991-1999) records reported by Hoagstrom (1994), Larson (1996), and Garrett (1997), along with unpublished data from the NMDGF/

U.S. Fish and Wildlife Service (FWS), and D.M. Hillis of UT-Austin, were compared to pre-1991 records.

This paper was primarily concerned with fish community surveys from the mainstem lower Pecos River, but species specific studies (e.g., Echelle and Echelle, 1978; Albeit, 1982; Humphries and Miller, 1982; Hatch et al., 1985; Kelsch and Hendricks, 1990) provided supplemental information. Additionally, studies focused on fishes of lower Pecos River tributaries (e.g., Stevenson and Buchanan, 1973; Kennedy, 1977; Cowley and Sublette, 1987; Propst, 1992) added insight for interpretation of historical and recent fish distributions.

RESULTS

Overall, the effort extended for recent collecting was similar to historical surveys of a given time period (Tables 2 and 3). Recent Carlsbad segment surveys were, in essence, a continuation of traditional NMDGF surveys (Propst, 1992; NMDGF and FWS unpublished data) with increased emphasis on native fishes such as *Cyprinodon pecosensis* (Echelle et al., 1997; Hoagstrom and Brooks, 1999) and *Cycleptus elongatus* (Propst, 1999). These efforts were similar to historical surveys, with omission of the fish barrier trap and surveys of Avalon and Red Bluff reservoirs, but with addition of boat-mounted electrofishing between Brantley Dam and Avalon Reservoir and within lower Carlsbad Lakes (Propst, 1992). Recent Toyah segment surveys were comparable to historical surveys (Table 2), but added gill-net sampling at a few locations (Hoagstrom, 1994; Larson, 1996). Recent Edwards segment surveys were least extensive compared to historical surveys (the main difference being lack of intensive sampling at two sites sensu Rhodes and Hubbs, 1992; Table 3). However, recent and historical data from the Edwards segment were considered at least marginally comparable, particularly because 1997 collections conducted by D.M. Hillis, UT-Austin, constituted a partial replication of his 1979 collections.

Forty-five native fish species have been reported from historical surveys of the lower Pecos River (not including the unsubstantiated *Phenacobius mirabilis* and *Ictiobus niger*; Table 1). Twenty-seven (60%) of these occurred in all three segments. Nine others (20%) were found in two different segments (3 in Carlsbad and Toyah, 5 in Toyah and Edwards, 1 in Carlsbad and Edwards). The remaining nine native fish species (20%) were only present in a single segment (3 in Carlsbad, 6 in Edwards). Total number of native species was 34, 35, and 39 in the Carlsbad, Toyah, and Edwards segments, respectively.

Recent collections included 30 native fish species (Table 1) with 14 historical inhabitants absent and one (*Ictalurus lupus*) uncertain, because of difficulties with identification (Yates et al., 1984; Kelsch and Hendricks, 1986). Only nine (30%) of the 30 remnant native species were found in all three segments, while seven (23%) were taken from two segments (2 in Carlsbad and Toyah, 3 in Toyah and Edwards, 2 in Carlsbad and Edwards). The remaining 14 native species (47%) were only present in collections from a single segment (5 in Carlsbad, 3 in Toyah, 6 in Edwards). The recent total of native species per segment was 18, 16, and 18 in Carlsbad, Toyah, and Edwards segments, respectively.

Table 2. Summary of fish community surveys within the lower Pecos River Toyah segment. Number of study sites and site visits is given for each survey and summarized for each of three survey periods (Post development period, 1939-1960; Near recent period, 1961-1990; Recent period, 1991-1999). Historical (1939-1990) collections were divided into two periods to provide a better comparison (i.e., even though this paper compares historical and recent collections, post-development and near recent fish communities were different, with post-development collections being more diverse. A comparison of effort from the three periods seems more informative than a comparison of collections from 51 years to those from 10). This table is believed to represent all major fish community surveys but may not represent all fish collections.

Survey	Surveys				Periods			
	Method	Sites	Visits	Period	Method	Sites	Visits	
UMMZ 1939-40	Bag seine	2	3 (1.5/site)	Post Development	Seine	21	50 (2.4/site)	
Jameson & Lindsay 1951	Bag and minnow seine?	1	1 (1.0/site)					
Hubbs & Springer 1954	Bag and minnow seine?	6	6 (1.0/site)					
Campbell 1959	Bag and minnow seine	21	40 (1.9/site)					
Surtkus et al. 1961-1976	Minnow seine	7	9 (1.3/site)	Near Recent	Seine	11	73 (6.6/site)	
Davis 1976	Minnow seine	2	2 (1.0/site)					
Hillis 1979	Minnow seine?	3	3 (1.0/site)					
Rhodes et al. 1987-88	Minnow seine	3	50 (16.7/site)					
Linam and Kleinsasser 1987	Bag and minnow seine	9	9 (1.0/site)	Recent	Seine	9	62 (6.9/site)	
Hoagstrom 1993-94	Minnow seine	6	50 (8.3/site)					
Larson 1994	Minnow seine	4	4 (1.0/site)					
Hoagstrom 1998	Minnow seine	8	8 (1.0/site)					
Hoagstrom 1993-94	Gill Net	5	6 (1.2/site)	Recent	Gill Net	9	10 (1.1/site)	
Larson 1994	Gill Net	4	4 (1.0/site)					

Table 3. Summary of fish community surveys within the lower Pecos River Edwards segment. Number of study sites and site visits is given for each survey and summarized for each of three survey periods (Post development period, 1939-1960; Near recent period, 1961-1990; Recent period, 1991-1999). Historical (1939-1990) collections were divided into two periods to provide a better comparison (i.e., even though this paper compares historical and recent collections, post-development and near recent fish communities were different, with post-development collections being more diverse. A comparison of effort from the three periods seems more informative than a comparison of collections from 51 years to those from 10). This table is believed to represent all major fish community surveys but may not represent all fish collections.

Survey	Surveys				Periods					
	Method	Sites	Visits	Period	Method	Sites	Visits			
Treviño-Robinson 1954	Bag and Minnow seine	1	1 (1.0/site)	Post Development	Seine	8	15 (1.9/site)			
Hubbs & Springer 1954	Bag and Minnow seine	2	2 (1.0/site)		Near Recent	Seine	18			
Campbell 1959	Bag and Minnow seine	7	12 (1.7/site)					Recent	Seine	11
Campbell 1959	Gill Net	3	3 (1.0/site)							
Suttkus et al. 1963-1976	Seine	2	12 (6.0/site)	Near Recent	Seine	18	69 (3.8/site)			
Hillis 1979	Minnow seine	17	17 (1.0/site)							
Garrett & Marsh 1980	Minnow seine	1	1 (1.0/site)	Recent	Seine	11	11 (1.0/site)			
Rhodes et al. 1987-88	Minnow seine	2	33 (16.5/site)							
Linam and Kleinsasser 1987	Bag and minnow seine	6	6 (1.0/site)							
Garrett 1991	Minnow seine/Electrofisher	1	1 (1.0/site)	Recent	Seine	11	11 (1.0/site)			
Larson 1994	Minnow seine/Gill Net	1	1 (1.0/site)							
Hillis 1997	Seine	9	9 (1.0/site)							

Table 4. Introduced fishes and native/nonnative hybrids known from the mainstem lower Pecos River including Amistad Reservoir, which inundates the Pecos River-Rio Grande confluence. Historical and recent distribution of each species is given by segment. ? = No locality information. Names and taxonomic order of North American freshwater fishes follow Mayden et al. (1992). Other names follow Robins et al. (1991).

Species	Historical Distribution	Recent Distribution
<i>Dorosoma petenense</i>	CARLSBAD	Carlsbad, uncommon
<i>Carassius auratus</i>	TOYAH - EDWARDS	-
<i>Ctenopharyngodon idella</i>	CARLSBAD	Carlsbad, Carlsbad Lake
<i>Cyprinella venusta</i>	TOYAH - EDWARDS	Edwards, common
<i>Cyprinus carpio</i>	ALL	All, abundant in upper segment
<i>Hybognathus placitus</i>	CARLSBAD	Carlsbad, rare
<i>Notemigonus crysoleucas</i>	CARLSBAD	Carlsbad, rare
<i>Notropis girardi</i>	CARLSBAD	-
<i>Catostomus commersoni</i> *	CARLSBAD	-
<i>Ameiurus melas</i>	ALL	Carlsbad, rare
<i>Ameiurus natalis</i>	TOYAH - EDWARDS	-
<i>Esox lucius</i>	EDWARDS	-
<i>Oncorhynchus mykiss</i>	CARLSBAD	-
<i>Menidia beryllina</i>	ALL	All, abundant in Carlsbad, common in Toyah, uncommon in Edwards
<i>Fundulus grandis</i>	ALL	All, common in Carlsbad & Toyah
<i>Cyprinodon pecosensis</i> x <i>C. variegatus</i>	ALL	All, abundant in Toyah, rare/localized elsewhere
<i>Gambusia geiseri</i>	EDWARDS	Edwards segment, uncommon
<i>Morone chrysops</i>	ALL	Carlsbad, rare
<i>Morone saxatilis</i>	CARLSBAD	-
<i>Morone saxatilis</i> x <i>M. chrysops</i>	CARLSBAD	-
<i>Ambloplites rupestris</i>	CARLSBAD	-
<i>Lepomis auritus</i>	EDWARDS	-
<i>Lepomis humilis</i>	?	-
<i>Lepomis microlophus</i>	EDWARDS	Edwards
<i>Micropterus dolomieu</i>	EDWARDS	-
<i>Micropterus punctulatus</i>	CARLSBAD	Carlsbad
<i>Pomoxis annularis</i>	ALL	Carlsbad
<i>Pomoxis nigromaculatus</i>	CARLSBAD	-
<i>Perca flavescens</i>	CARLSBAD	-
<i>Stizostedion canadense</i>	EDWARDS	-
<i>Stizostedion vitreum</i>	CARLSBAD	Carlsbad, rare
<i>Cynoscion nebulosus</i>	CARLSBAD	-
<i>Micropogonias undulatus</i>	CARLSBAD	-
<i>Pogonias cromis</i>	CARLSBAD	-
<i>Sciaenops ocellatus</i>	CARLSBAD - TOYAH	-
<i>Paralichthys lethostigma</i>	CARLSBAD	-

*Native to Pecos River headwaters, presumed introduced to the lower Pecos River.

Ten native species appeared to be thriving or have stable populations during recent collections (Table 1). Status of the remaining 20 is either diminished or tenuous (diminished species were sporadic in occurrence and/or restricted in distribution; tenuous species were very rare). *Macrhybopsis a. aestivalis*, *Notropis l. ludibundus*, and *Aplodinotus grunniens* were each represented by a single individual. The number of native

species missing from recent collections was 16, 19, and 20 per segment (Carlsbad, Toyah, and Edwards respectively), representing fish species richness reductions of 47, 54, and 51% respectively.

Thirty-six introduced fish species were reported from historical lower Pecos River surveys (Table 4). Seven of these (19%) were known from all segments

and four (11%) were found in two segments (1 in Carlsbad and Toyah, 2 in Toyah and Edwards). Twenty-four introduced species (69%) were restricted to a single segment (18 to Carlsbad, 6 to Edwards). Total introduced species per segment was 26, 11, and 16 (Carlsbad, Toyah, and Edwards, respectively). Historical distribution of *Lepomis humilis* was not reported (Campbell, 1959).

Sixteen introduced fish species were present in recent collections (Table 4). Four (25%) were found

in all segments, but the remaining 12 (75%) were restricted to a single segment (9 to Carlsbad, 3 to Edwards). Five of these (*Hybognathus placitus*, *Notemigonus crysoleucas*, *Ameiurus melas*, *Morone chrysops*, *Stizostedion vitreum*) were rare, likely representing bait bucket releases (first two) or strays from reservoirs or tributaries (last three). The recent total of introduced species per segment was 13, 4, and 7 (Carlsbad, Toyah, and Edwards, respectively).

DISCUSSION

Comparison of historical and recent native fish species composition indicated significant decline. Historically, the three segments had similar fish species richness and pre-dam (i.e., pre-systematic fish survey) similarity between segments was likely even greater. The lower Pecos River did not sustain a commercial fishery (as did large rivers elsewhere in North America) so the public may have been relatively unaware of what fishes were present, causing large-river fishes (e.g., *Atractosteus spatula*, *Lepisosteus oculatus*, *Anguilla rostrata*, *C. elongatus*, and *A. grunniens*) to be poorly documented in historical accounts. For example, Hubbs (1957) believed *Lepisosteus platostomus*, collected by Pope in 1854 (see Evermann and Kendall, 1894) represented *A. spatula*, whereas Sublette et al. (1990) suggested the species captured was *L. oculatus*. Following the first development period (after 1930), *L. oculatus* occurred further upstream than *A. spatula* (Campbell, 1959; J. F. Scudday, Sul Ross State University, pers. comm.), suggesting Pope's specimen was more likely *L. oculatus*, but human-caused changes in flow regime, channel sediment, and water quality made the middle Twentieth Century lower Pecos River much different from that of 1854 (see above). Thus, it is possible both gar species were present in New Mexico prior to impoundment, neither being documented.

Researchers active throughout the second development period noted the decline of lower Pecos River fishes. During a 1947 visit to Malaga Bend, Koster complained, "Collecting to date has been disappointing. Fish are scarce. Many species which are known from both above and below are seemingly ab-

sent from this lower stretch of the Pecos" (from field notes 1939-1955). The dramatic decline of *Notropis* species serves as an example. *Notropis jemezianus*, once widespread (Campbell, 1959; Treviño-Robinson, 1955; Sublette et al., 1990), was absent from recent collections (possibly persisting in Independence Creek [Garrett, 1997; Karges, The Nature Conservancy, pers. comm.]). *Notropis l. ludibundus* disappeared from the Carlsbad segment before 1975 (Sublette, 1975) and is known from a single recent Toyah segment specimen (Hoagstrom, 1994). *Notropis amabilis* and *N. braytoni*, historically found as far upstream as Roswell, New Mexico (Hatch, 1985; Platania, 1996), are rare in recent Toyah and Edwards segment collections. *Notropis girardi*, introduced to the Carlsbad segment around 1978 (Bestgen et al., 1989), was apparently never established.

Notropis simus pecosensis now is restricted to the middle Pecos River between Sumner and Brantley dams (Sublette et al., 1990; Propst, 1999), but the nominal collection of this subspecies was made in 1854 by Pope (Chernoff et al., 1982) who surveyed the Pecos River from Black River confluence, downstream to Emigrant Crossing near Barstow, Texas (Pope, 1854). Although the exact location of Pope's *N. s. pecosensis* collection was unspecified (Evermann and Kendall, 1894; Platania, 1995), this record indicates that *N. s. pecosensis* historically inhabited the lower Pecos River when it was deep and swift, with a sand bed. By the time subsequent fish surveys were conducted, the lower Pecos River had become salty, sluggish, and silty.

Notropis orca and *N. buchanani* were rare in historical lower Pecos River collections, but common in the adjacent Rio Grande (Treviño-Robinson, 1955, 1959; Chernoff et al., 1982). Their abundance in the pre-impoundment lower Pecos River is unknown, but their disappearance from the drainage may have foreshadowed their decline in the Rio Grande. *Notropis orca* is now extinct (Chernoff et al., 1982; Bestgen and Platania, 1990; Hubbs et al., 1991), and *N. buchanani* is rare within the Rio Grande (Platania, 1990; Edwards and Contreras-Balderas, 1991). Thus, the more recent decline of *N. braytoni* and *N. jemezianus* from the lower Pecos River may justify increased concern for Rio Grande populations of these species.

Similar to developments for irrigated agriculture (Taylor, 1902; President's Water Resources Policy Commission, 1950; Lingle and Linford, 1961), attempts to establish productive sport fisheries in the lower Pecos River did not meet expectations (Campbell, 1959; Navarre, 1959, 1960; Little, 1961b, 1963a, 1963b, 1964c, 1964d, 1965). Because game-fish management efforts were largely initiated subsequent to major water development, conditions that prevented fishery success were not solely attributable to geological or human-induced factors, but represented a combination of both (Campbell, 1959; Little, 1964c; Davis, 1987). Poor and unstable water quality (Little, 1964d; Larson, 1996; Linam and Kleinsasser, 1996) and toxic algal blooms (Rhodes and Hubbs, 1992) significantly impacted game fish success. As a result of dramatic human-caused changes, the historical status of native game fishes and potential of the pre-impoundment lower Pecos River to support nonnatives will never be known.

Some native species present in historical collections may not have been established within the pre-impoundment lower Pecos River. For example, *Rhinichthys cataractae*, was common in the upper Pecos River, but infrequent in the lower Pecos River (Miller, 1977; Sublette et al., 1990). This species may have colonized tailwaters of McMillan and Avalon reservoirs during floods or with human aid. Both tailwater reaches were eventually dewatered, so absence of *R. cataractae* from recent collections is not surprising.

Etheostoma lepidum typically inhabits small streams with dense vegetation (Hubbs et al., 1953; Cowley and Sublette, 1987) and it was never abun-

dant in the mainstem Pecos River (Hubbs and Echelle, 1972). The species may have entered the Pecos River from tributaries, colonizing spring-fed areas that dominated the mainstem after the first development period. These darters probably declined during the second development period as spring flows were depleted. Similarly, Balconian fishes (*Campostoma anomalum*, *Cyprinella proserpina*, *Dionda episcopa*, *N. amabilis*, *N. l. ludibundus*, *Pimephales vigilax*, *Scartomyzon congestus*, *Etheostoma grahami*) and Tamaulipan fishes (*N. braytoni*, *Cichlasoma cyanoguttatum*) were most abundant in spring-fed tributaries (Rhodes and Hubbs, 1992) and their decline from the lower Pecos River coincided with spring flow depletion (Linam and Kleinsasser, 1996).

Cyprinodon pecosensis and *Fundulus zebrinus* were probably uncommon in the pre-impoundment Pecos River mainstem, most likely occupying saline tributaries and floodplain wetlands (Hoagstrom and Brooks, 1999). Both species proliferated in the lower Pecos River as it was dewatered (Campbell, 1959), but loss of floodplain wetlands eventually restricted them to the mainstem and a few persistent tributaries (Hoagstrom and Brooks, 1999). Subsequently, mainstem populations were decimated by introduced congeners. A *C. pecosensis* x *C. variegatus* hybrid swarm replaced *C. pecosensis* (Echelle et al., 1987; Echelle and Connor, 1989; Wilde and Echelle, 1992), and *F. zebrinus* was replaced by *F. grandis* (Hoagstrom, 1994). *Cyprinodon pecosensis* (3 locations) and *F. zebrinus* (3 locations) persist in off-channel locations, with the largest populations of both species inhabiting upper Salt Creek, Culberson and Reeves counties, Texas (N.L. Allan, FWS; A.A. Echelle, Oklahoma State University; G.P. Garrett, TPW; J.P. Karges, The Nature Conservancy, pers. comm.).

In each lower Pecos River segment, recent native fish species richness was roughly half of historical richness. Resultant fish communities represented a response to the water quality, physical habitat, and flow regime of each segment. Differences among segments were exacerbated by toxic algal blooms and physical barriers. Native species persistent in the Carlsbad segment included lentic freshwater fauna (e.g., *Dorosoma cepedianum*, *Lepisosteus osseus*, centrarchids), generalist freshwater fauna (e.g., *Cyprinella lutrensis*, *Pimephales promelas*, *Menidia*

beryllina, *Gambusia affinis*), and riverine catostomids (e.g., *Carpionodes carpio elongatus*, *C. elongatus*, *Ictiobus bubalus*). The Carlsbad segment was heavily impacted by impoundment, agriculture, and urbanization, with significant impacts from diversion, oil field pollution, groundwater pumping, and upstream development (U.S. National Resources Planning Board, 1942; Thomas et al., 1963; Davis, 1987), readily accounting for recent absence of 16 native species. Relatively high recent introduced species richness (n=13) is attributable to presence of three large reservoirs, absence of toxic algal blooms (except in Red Bluff Reservoir), and persistence of spring flows in the city of Carlsbad, Black River, and Delaware River.

Euryhaline fishes (e.g., *D. cepedianum*, *C. pecosensis* x *C. variegatus*, *F. grandis*, *Lucania parva*, *G. affinis*) dominated the Toyah segment, which was not surprising in light of numerous impacts that concentrated salts therein, including upstream development (e.g., Carlsbad segment), local groundwater withdrawal, oil field pollution, mainstem and tributary diversion, and agriculture (Taylor, 1902; U.S. National Resources Planning Board, 1942; Grozier et al., 1966; Mace et al., 2001). Toxic algal blooms further impacted Toyah segment fishes (Rhodes and Hubbs, 1992; Hoagstrom, 1994). Failure of introduced game fishes (none taken in recent surveys) is attributable to absence of large reservoirs, absence of significant spring inflows, and toxic algal blooms. Severity of direct and indirect impacts on the Toyah segment and absence of redeeming habitat features (e.g., persistent springs) clearly account for recent absence of 19 native species.

The Edwards segment supported freshwater generalists (e.g., *C. lutrensis*, *C. venusta*, *P. vigilax*, *G. affinis*) and spring-dwelling specialists (e.g., *C. proserpina*, *D. episcopa*, *E. grahami*). It is possible that the Edwards segment retains species not taken in recent surveys, because recent surveys were not extensive (Table 3) and did not include gill-net or boat-mounted electrofishing collections, increasing the likelihood that riverine species (e.g., *L. oculatus*, *C. elongatus*, *I. bubalus*, *I. furcatus*, *A. grunniens*) could have escaped detection. Even so, severe upstream impacts (e.g., Carlsbad and Toyah segments) could account for the recent absence of 20 native species,

especially in combination with toxic algal blooms that reduce short-term species richness and possibly cause long-term reductions (Rhodes and Hubbs, 1992). Downstream impoundment of the Edwards segment by Amistad Reservoir could also have facilitated species loss (Winston et al., 1991; Wilde and Ostrand, 1999; Lienesch et al., 2000). Persistence of nonnative game fishes in the Edwards segment may be attributed to persistent spring flows and Amistad Reservoir.

Minckley (1965) suggested that the lower Pecos River was incidentally stocked with *M. beryllina* along with estuarine sport-fishes (e.g., *Sciaenops ocellatus*). Subsequently, *M. beryllina* dispersed beyond the lower Pecos River (Hubbs and Echelle, 1972), currently ranging as far as 188 km upstream of Brantley Dam (New Mexico Fishery Resources Office, FWS, unpublished data). Incidental stockings of this sort probably introduced estuarine invertebrates (Davis, 1987) and could have played a role in establishing *C. variegatus* (*C. variegatus* were reported by Campbell in 1959 in the same time period and vicinity of *S. ocellatus* introductions, but no voucher specimens are available).

Successful introduction of *Cyprinodon variegatus* to the lower Pecos River most likely occurred after 1980, with establishment and spread facilitated by bait-bucket release (Echelle et al., 1987; Echelle and Connor, 1989; Hubbs et al., 1991; Wilde and Echelle, 1992; Echelle et al., 1997; Echelle et al., this volume). Genetic evidence suggests the species first colonized Red Bluff Reservoir via introduction from Lake Balmorhea, Reeves County, Texas (Childs et al., 1996). The Lake Balmorhea *C. variegatus* population was established from an unknown source prior to 1968 (Stevenson and Buchanan, 1973) and persists today (Echelle et al., this volume). It also served as the source for a recent *C. variegatus* introduction to Diamond Y Spring (Echelle and Echelle, 1997). The point of introduction for *F. grandis* has not been specifically investigated, but establishment in the Edwards segment (Hillis et al., 1980; Hubbs, 1982; Rhodes and Hubbs, 1992; Linam and Kleinsasser, 1996) and Carlsbad segment (Propst, 1992), prior to expansion into Toyah segment (Hoagstrom, 1994; Larson, 1996) suggests at least two separate introductions.

CONCLUSIONS

The historical fish fauna of the lower Pecos River included many riverine forms and was not fragmented by physical barriers. Early Anglo-American development (1885 to 1929) resulted in capture and diversion of surface waters and alluvial sediments, after which river substrate changed and local groundwater springs became the primary source of river flow. A second phase of Anglo-American development (1930 to 1970) reduced inflow from groundwater, in many cases directing groundwater flow away from the river, while additional dams furthered river fragmentation. As a result, lower Pecos River hydrology, geomorphology, and water chemistry were dramatically altered from a natural condition that was never quantitatively described.

In response to human induced changes, each river segment developed a distinctive fish community composed of tolerant native and introduced fishes. Native riverine fishes declined from all segments because mainstream habitats were universally impacted. Spring and wetland fishes retreated to areas sustaining substantial spring inflows (e.g., Black River, Salt Creek, Independence Creek, lower Pecos River Edwards segment). The Toyah segment, was most dramatically impacted by development. As a result, this segment suffered the greatest percent native species richness reduction and did not sustain introduced game fishes.

Intentional game fish stockings met with only short-term success while incidental stocking and bait-bucket release established nonnative euryhaline fishes. Success of game species was likely limited by unfav-

orable water quality, degraded habitat, and toxic algal blooms. These same factors apparently favored unintentionally introduced euryhaline fishes. Additional nonnative fish introductions and spread of locally established nonnatives continue to threaten native lower Pecos River fishes.

Because of numerous, large-scale impacts that have altered the mainstem lower Pecos River, challenges for native fish conservation are many and great. Proponents of aquatic habitat restoration and native fish protection will benefit from an appreciation of former fish community diversity and severity of impacts that changed the lower Pecos River. While it is appropriate to consider the entire lower Pecos River as a significant component of the historical range of many native Rio Grande basin fishes, it may be optimistic to expect restoration of all or even a few missing species (in light of prevailing conditions). However, a number of persistent native species (including unique Pecos River and Rio Grande forms, Table 1) would benefit from immediate population assessment and conservation activity. Improvement of conditions to preserve these species may also facilitate voluntary re-establishment of fishes occupying the adjacent Rio Grande, Pecos River tributaries, or the middle Pecos River. At least, recognition of rapid and ongoing decimation of native lower Pecos River fishes should increase awareness of the general imperilment of Rio Grande basin fishes, focusing attention on and elevating prioritization of waters where remnant native fish assemblages persist.

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PUPFISHES OF THE NORTHERN CHIHUAHUAN DESERT: STATUS AND CONSERVATION

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ABSTRACT

Twelve species of pupfish (Genus *Cyprinodon*) generally are recognized in the northern Chihuahuan Desert. Eight of these are restricted to relatively small spring systems, whereas the remaining four occur in springs and riverine situations. The present abundance and distribution of pupfishes in the region is only a remnant of what must have been present prior to anthropogenic watershed deterioration and depletion of groundwater. Today, most spring-dwelling pupfishes are succumbing to losses of springflows, primarily as a result of pumping of groundwater, and the riverine

species are adversely affected by a diversity of anthropogenic factors. The diversity of both groups has declined as a result of introgressive hybridization with a non-native pupfish, the wide ranging coastal species *C. variegatus*. The rapidity with which native stocks can be lost as a result of such hybridization is dramatically illustrated by events following the introduction of *C. variegatus* into the Pecos River Basin in the 1960s. A similar threat is posed by transport of any non-native pupfish into waters occupied by an endemic pupfish.

INTRODUCTION

In this paper, we review the history, current status, and conservation efforts for the 12 species of pupfish (Cyprinodontidae: *Cyprinodon*) generally recognized in the northern Chihuahuan Desert. The genus ranges west to east from the Death Valley System to the West Indies and north to south in coastal waters from Cape Cod, Massachusetts to Venezuela. In the past century, seven of the 25 species of *Cyprinodon* in the arid southwest have been driven to extinction in the wild. The first of these events occurred sometime between 1903 and 1953 when *C. latifasciatus* of the Parras Basin in Coahuila disappeared as a result of a variety of anthropogenic factors, including destruction of springs (Miller, 1964). Most recent extinctions occurred in the 1980s and early 1990s with the loss of five *Cyprinodon* species in the Sandia and Potosí basins of southern Nuevo Leon, all as a result of groundwater pumping and resultant loss of springs (Contreras-B and Lozano-V, 1996), including a complex of small to large springs that was unknown to ichthyologists until 1983 (Lozano-V and Contreras-B, 1993). The

seventh known species of *Cyprinodon* to go extinct in historic times is the Monkey Springs pupfish, which was extirpated as a result of groundwater pumping and other human activities (Minckley, 1973; Minckley et al., 2002). Although extinct in the wild, three of the seven species are being maintained in various aquarium facilities (Lozano-V and Contreras-B, 1993).

In the following accounts, we present a basin-by-basin account of the pupfishes in the northern Chihuahuan Desert, eight of which are restricted to single springs or spring systems, whereas the other four occur in riverine situations and associated springfed waters. Regardless of extent of range, however, the pupfishes of the northern Chihuahuan Desert are declining. This reflects a variety of anthropogenic factors, but loss of springs, apparently as a result of groundwater pumping, and genetic introgression by non-native pupfish are the most immediate causes of concern.

PECOS RIVER BASIN

Cyprinodon bovinus (Leon Springs pupfish).—The waters supporting *C. bovinus* occur in Diamond Y Draw (= Leon Creek in some publications), a “flood” tributary of the Pecos River that has rarely, if ever, reached the river in historic times. Until 1965, the species was unknown to science except for the original collection, in 1851, of 16 specimens from Leon Springs near the present Fort Stockton, Pecos County, Texas. In 1965, W. L. Minckley found it approximately 15 km downstream of Leon Springs. In the intervening time, an area immediately downstream of, and fed by, Leon Springs had been dammed (about 1910) and the resulting “Lake Leon” was stocked with carp and game fish. In 1938, Carl Hubbs failed to collect the species in the Leon Springs area. By 1958, Leon Springs had gone dry because of over-pumping of the aquifer (Brune, 1975). The species was declared extinct by the late 1950s (Hubbs, 1957; Miller, 1961). Its rediscovery was verified in a re-description of the species (Echelle and Miller, 1974), and it was listed as a federally endangered species in 1980, with most of the occupied areas designated as critical habitat (Federal Register, 45:54678). Since 1990, most of the area occupied by *C. bovinus* has received protection as a preserve of 607 hectares (Diamond Y Spring Preserve) managed by The Nature Conservancy of Texas.

Since its re-discovery, *C. bovinus* has occupied two separate systems of surface water separated by 1-2 km of dry land: an “upstream watercourse” receiving flow from Diamond Y Spring and several smaller springs and seeps, and a “downstream watercourse” receiving flow from Euphrasia Spring, several small seepage springs, and groundwater seepage. Both of the two watercourses are 2-4 km long, with length of watercourse varying considerably over the years.

The primary threats to *C. bovinus* include pollution, loss of habitat, and introgressive hybridization with an introduced congener. Both watercourses supporting *C. bovinus* occur in the Fort Stockton Oil and Gas Field, an area of intense petrochemical extraction activity for about 50 years. Thus, there is a continuing threat of pollution (Kennedy, 1977; Gehlbach, 1981). In 1974, the Soil Conservation Service (Natural Re-

sources Conservation Service) constructed an earthen berm that protects the headpool of Diamond Y Spring from surface spills (Hubbs 1980). In 1992, a ruptured pipeline released crude oil in a nearby area, creating sufficient concern that the oil company dug a trench downslope of the spill to help slow contamination of the watercourse.

In recent years, both watercourses supporting *C. bovinus* have been 1-2 km shorter than they were in the 1970s (Hubbs et al., 1978; Echelle and Echelle, 1980). This may be the combined result of the drought conditions in the region during the past five years and continued over-pumping of groundwater in the basin. In addition to the reduced size of the watercourses, densities of pupfish now seem lower than they were in the past. In the 1970s, the pupfish was abundant in a diversity of open-water situations in Diamond Y Draw (Kennedy, 1977; Hubbs et al., 1978; Echelle and Echelle, 1980). Reduced densities seem to reflect a loss of open-water habitat as a result of encroachment by bulrush because of reduced water flow.

The problems posed by introductions of non-native pupfish emerged as the most important immediate threat to *C. bovinus* shortly after R. D. Suttkus collected *C. variegatus* from the lower watercourse in 1974. By January 1976, hybrid morphotypes occurred throughout the lower watercourse. This led to an intensive effort to eliminate hybrids, with some attention to protecting the invertebrate community (Hubbs et al., 1978). The effort included treatment of the lower watercourse with rotenone, re-introducing *C. bovinus* from the upper watercourse, and subsequent seining and selective removal of suspected hybrids (Hubbs, 1980). The absence of morphological traits indicating hybrids (Hubbs, 1980) and the absence of alleles of *C. variegatus* in a genetic survey of the population (Echelle et al., 1987) indicated that the renovation was successful in restoring *C. bovinus* to the lower watercourse (for a more detailed review see Minckley et al., 1991). In retrospect, it was extremely fortunate that, during the 1976 renovation, *C. bovinus* from the upper watercourse was used to establish a captive stock at Dexter National Fish Hatchery and Technology Center in New Mexico (DNFH).

A second introduction of *C. variegatus* into Diamond Y Draw in the late 1980s or early 1990s led to contamination of both the upper and the lower watercourse of Diamond Y Draw (Echelle and Echelle, 1997). In response, the U.S. Fish and Wildlife Service (FWS) approved and, with help from various agencies and the Rio Grande Fishes Recovery Team, implemented a plan to restore the native pupfish genome. The renovation occurred in two phases that differed in approach. Antimycin A was used to eliminate all fish (including two non-natives, *Gambusia geiseri* and *C. carpio*) from the Diamond Y Spring outflow in August 1998. Prior to the renovation, large samples of the native fishes *G. nobilis* and *L. parva* were removed and transported alive to DNFH; after dissipation of the toxin, they were released back into the watercourse. Similar precautions were taken to protect known invertebrate species of concern. To provide additional protection for the remainder of the fauna, two small areas supporting pupfish were left untreated; one of these was contaminated by *C. variegatus*, but the population seemed sufficiently small that it was decided to try diluting the introgressed genome by releases of pure *C. bovinus* from DNFH. Renovation of the downstream watercourse was initiated in March 2000 and involved removal of all pupfish captured by intensive seining, dip-netting, and trapping. Both watercourses received large numbers of pupfish from the captive DNFH stock of *C. bovinus* immediately after renovation and in the following year.

Subsequent genetic surveys indicated that the renovation efforts were largely successful in both watercourses (AAE and AFE, unpublished data). There was no evidence of introgression in the upper watercourse except for the small population not treated with ichthyotoxins. Levels of introgression in the lower watercourse were reduced to possibly acceptable levels. Further releases of *C. bovinus* from the DNFH stock are planned for the future, with emphasis on the known areas of persistent contamination.

Since establishing the Diamond Y Preserve, The Texas Nature Conservancy (TNC) has attempted to restore the watershed to more natural conditions. Their activity includes renovation of old oil/gas pads, management of livestock to reduce their impact on the aquatic system, and using tractor equipment in 2000 to uproot all salt cedars (*Tamarix* sp.) in the lower

watercourse. TNC is searching for appropriate ways to improve conditions for the pupfish without compromising the habitat for other rare, native species.

Cyprinodon pecosensis (Pecos pupfish).—The geographic range of *C. pecosensis* once included saline floodplains and springs, lakes, gypsum sinkholes, and other waters associated with the Pecos River from Bitter Lake National Wildlife Refuge and Bottomless Lakes State Park near Roswell, New Mexico, downstream for about 650 river-km to the mouth of Independence Creek in Texas (Echelle and Echelle, 1978). The present distribution represents less than 20% of the historic range. The only known natural population remaining in Texas (but see below) is in a portion of Salt Creek, a saline tributary of the Pecos River just south of the New Mexico border. In New Mexico, the species occurs most abundantly in saline waters of the Pecos River floodplain near Roswell, primarily on the Bitter Lake National Wildlife Refuge and Bottomless Lakes State Park.

There is only sketchy knowledge of the distribution and abundance of Pecos pupfish prior to the extensive habitat alteration that had occurred by 1950, but anthropogenic factors have undoubtedly caused a considerable loss of habitat for the species. The factors contributing to such losses were reviewed by Hoagstrom and Brooks (1999). Native Americans used water from the Pecos River for agriculture in the headwaters of the river prior to 1600, but human-induced alterations of aquatic habitats in the basin probably escalated considerably in the late 1800s. Since that time, a variety of anthropogenic factors drastically reduced the habitat available to pupfish. These include construction of dams on the mainstem of the river and tributary streams, introduction of salt cedar, over-pumping of aquifers, over-grazing by domestic animals, erosion, pollution, and draining of wetlands (Hoagstrom, this volume). An apparently indirect effect of human activity are fish kills that have occurred sporadically in the lower Pecos River in Texas since the 1950s, including several kills from 1985 to 1989 that extended over hundreds of river kilometers (Rhodes and Hubbs, 1992; Childs et al., 1996). The latter series of kills apparently resulted from toxins released during blooms of a chrysophyte alga (*Prymnesium parvum*) that probably reflect a response to anthropogenic nutrient enrichment (Rhodes and

Hubbs, 1992). Although not documented in detail, such kills undoubtedly resulted in temporary depletions of the pupfish population.

Until recently, and despite losses of habitat and extensive fish kills, *C. pecosensis* seemed reasonably secure because of its relatively large range and locally high abundances. However, the status of the species changed abruptly with the introduction, sometime between 1980 and 1984 (Echelle and Connor, 1989), of *C. variegatus* into the basin, probably in Red Bluff Reservoir on the New Mexico/Texas boundary (Childs et al., 1996). Collections of pupfish from the Pecos River at four Texas localities in March 1980 showed no morphological evidence of influence by *C. variegatus*. But less than five years later, in August 1984, collections of six specimens each from two sites separated by about 200 river-km comprised *C. pecosensis* x *variegatus* hybrids (Echelle et al., 1987). The genotypes for allozyme loci indicated that samples consisted of individuals that were minimally second-generation hybrids.

A broader geographic survey in 1985 detected locally panmictic hybrid populations throughout approximately 430 river-km of the Pecos River in Texas, where, depending on locality, alleles typical of *C. variegatus* represented 18 to 84 percent of the genome (Echelle and Conner, 1989). Subsequent monitoring revealed that hybrids were ubiquitous in the Pecos River and peripheral waters (reservoirs, irrigation canals, and gravel pits) in an area that extended downstream from near Loving, New Mexico to at least the vicinity of Pandale, Texas, approximately 55 km downstream of the historic range of *C. pecosensis* (Wilde and Echelle, 1992, 1997; Echelle et al., 1997; AAE and AFE, unpubl. data). Childs et al. (1996) suggested that the genetic structure of the hybrid swarm is explained by genetic swamping, possibly mediated by selection for *C. variegatus* or *C. pecosensis* x *variegatus* hybrids during a period of increasing population size, such as those that would have followed the fish kills mentioned earlier in this account. The role of selection is being confirmed experimentally by J. Rosenfeld (pers. comm.) in A. Kodric-Brown's laboratory at the University of New Mexico.

Until recently, the population in Salt Creek, a tributary of the Pecos River near the New Mexico/Texas

boundary, was considered effectively free of contamination except near the mouth of the creek where there were low frequencies of alleles typical of *C. variegatus*. However, in March 2001, morphological and genetic evidence of hybrids extended approximately 19 river-km upstream in Salt Creek (AAE and AFE, unpubl.). Pure populations of Texas stocks of *C. pecosensis* exist only in two artificial ponds supporting stocks transplanted from Salt Creek (Garrett, this volume) and, as of March 2001, in a headwater reach of Salt Creek (AAE and AFE, unpubl. data).

Endangered species status was proposed for *C. pecosensis* in 1998, primarily because of the threat from hybridization (Federal Register, 63:4608). In 2000, the proposal was withdrawn (Federal Register 65:14513) in response to a Conservation Agreement between the states of New Mexico and Texas, the FWS, and the U.S. Bureau of Land Management (Federal Register, 65:71424). In this agreement, "The signatory agencies . . . made commitments to protect known extant populations of pure Pecos pupfish, expand the distribution of the species within its native range by establishing new populations, and to prohibit the use of . . ." *C. variegatus* as baitfish in the Pecos River area. To date, various measures have been taken to protect populations, including, among other proactive measures, constructing fish barriers in two locations, initiating a study of the life history of the pupfish, enacting the necessary baitfish regulations, and establishment of two captive populations of the Texas stock in artificial ponds (Garrett, this volume).

Cyprinodon elegans (Comanche Springs pupfish).—This pupfish is known only from springfed systems in two separate flood tributaries of the Pecos River in Trans-Pecos Texas, Comanche Draw and the Toyah Creek Basin. Unlike the other two pupfishes endemic to the Pecos River drainage, which occur in saline to moderately saline waters, *C. elegans* is known only from relatively fresh waters (about 1-3 ppt total dissolved solids). The species was described from 32 specimens taken in 1851 at Comanche Springs, Fort Stockton, Texas. Because of over-pumping of groundwater, the six large springs of the Comanche Springs complex were dry by 1956 (Brune, 1981; Scudday, this volume) and the morphologically divergent (Echelle, 1975) local population of *C. elegans* was extirpated (Hubbs, 1957; Miller, 1961).

The species still exists about 90 km to the west in the Toyah Creek Basin near Balmorhea and Toyahvale, Reeves and Jeff Davis counties, where there is a system of three large artesian springs (Phantom Lake, San Solomon, and Giffin springs) and smaller springs that feed the irrigation canals for Reeves County Water Improvement District No. 1 (Garrett and Price, 1993; Garrett, this volume). Genetic and morphological studies indicate that the Phantom Lake Spring population is divergent from the populations in waters fed by Giffin and San Solomon Springs (Echelle, 1975; Echelle et al., 1987).

The species occurs in a small segment (< 1 km) of Toyah Creek, a large, swimming pool fed by San Solomon Spring at Balmorhea State Park, and two semi-natural refugia at the park. The remainder of the species is almost entirely confined to a system of earthen and concrete irrigation canals that is fed primarily by the artesian springs and serves approximately 2428 ha of agricultural land (LaFave and Sharp, 1987). During cooler months of the year, much of the flow is diverted into Lake Balmorhea, an artificial reservoir. The canal system supporting *C. elegans* extends through an area about 3 to 4 km wide and 15 km long, but the species is primarily restricted to areas of more permanent water in the main canals, in sections where the current is slower and the substrate more heterogeneous.

The springs now supporting *C. elegans* originally would have fed large, marshy habitats (ciénegas) that drained into Toyah Creek. However, such ciénegas would have been eliminated by the development of the system of irrigation canals for agriculture (Garrett, this volume). Traces of canals built by Native Americans occur in the vicinity of San Solomon Springs (Brune, 1975, 1981), but large-scale diversion of springflows probably started in the 1870s when the area was developed for agriculture to supply the military at Fort Davis (Young et al., 1993). Such activity culminated with the amalgamation, in 1914, of local canal companies into the Reeves County Water Improvement District No.1 and reconstruction of the canal system by the U.S. Bureau of Reclamation in 1946 (Young et al., 1993). A variety of other fishes are known from the area, including at least eight introduced species, most of which are restricted primarily to Lake Balmorhea where they were released either to support the sport fishery or as accidents associated with release of

gamefishes. Since the 1960s, Lake Balmorhea has supported a dense, non-native population of *Cyprinodon variegatus*, probably as a result of accidental transport (Stevenson and Buchanan, 1973).

The presence of an introduced population of *C. variegatus* in Lake Balmorhea threatens existing populations of *C. elegans* with both hybridization and competition for resources. A hybrid zone between the two occurs in an earthen canal where Lake Balmorhea receives flow from the irrigation system (Stevenson and Buchanan, 1973; A. F. Echelle and Echelle, 1994). Upstream migration of *C. variegatus* from the reservoir into most of the spring system is precluded by physical barriers. However, in the summer of 1988, *C. variegatus* had moved by way of a recently dug canal from the vicinity of the lake into East Sandia Spring, a small spring near the periphery of the spring-system in the region. A large sample of the introduced species was taken from the headpool of that spring in July 1988 (A. F. Echelle and Echelle, 1994), but sometime after that, and for unknown reasons, *C. variegatus* disappeared from the spring. To our knowledge there is no historical record of *C. elegans* from the headpool of East Sandia Spring, but on 20 March 2001 two specimens were found just downstream of a small culvert in the outflow. There is some indication of male sterility among hybrids in Lake Balmorhea, but the presence of backcross progeny demonstrates that genetic introgression by *C. variegatus* is a serious concern should *C. variegatus* gain access to areas outside of the lake (A.F. Echelle and Echelle, 1994). In 1998, Texas Parks and Wildlife Department expended considerable effort to eliminate *C. variegatus* from the lake; although more than 5 million *C. variegatus* were initially eliminated, the fish quickly reestablished in the lake (Garrett, this volume).

The ultimate threat to *C. elegans* is habitat loss due to declining springflows. Periodic losses of pupfish due to management of the irrigation canals supporting the species (Davis, 1979) is a relatively trivial matter. Some springs in the Balmorhea-Toyahvale area have gone dry and flows from all springs in the area have declined since the early 1900s (Brune, 1981; A. F. Echelle et al., 1989). The two largest of the existing springs, Phantom Lake and San Solomon springs, showed a steady decline in recent decades (A. F. Echelle et al., 1989; Schuster, 1997) and surface flow from the former ceased in 2000 (N. Allan, pers. comm.). In

2001, the Bureau of Reclamation and the U.S. Fish and Wildlife Service installed a pump to maintain the small pool at the head of Phantom Lake Spring, but this is a short-term solution that is unlikely to support the fish populations if water levels continue to decline (N. Allan, FWS, pers. comm.).

Cyprinodon elegans has been a federally listed endangered species since 11 March 1967 (Federal Register, 32:4001). Three aquatic refugia have been constructed for *C. elegans* and other endemic forms. Two using flows from San Solomon Spring were constructed at Balmorhea State Park by the Texas Parks and Wildlife Department; one, constructed in 1974, is a meandering, slow-flowing channel about 120 m long (Echelle and Hubbs, 1978) and the other is an artificial, 1-ha ciénega constructed in 1996 (Garrett, this volume). A refugium at Phantom Lake Spring was built in 1993 through the cooperation of several state and

federal agencies. For this refugium, some of the water emerging from Phantom Cave, a hillside cavern, is diverted from an irrigation canal into an artificially constructed channel and side-pool habitat that is about 110 m long (Young et al., 1993). In the Phantom Lake Spring refugium, the abundance of the pupfish peaked rapidly in the first year and then declined somewhat with increasing growth of vegetation, possibly as a result of the elimination of open patches of bottom substrate for spawning (Winemiller and Anderson, 1997). This refugium has been ineffective since 1999 as a result of the loss of springflows. However, the two refugia at Balmorhea State Park support large populations of *C. elegans* in semi-natural settings (Garrett, this volume). Captive stocks of the Phantom Lake Spring population have been maintained at the Uvalde National Fish Hatchery in south-central Texas since 1990.

RÍO CONCHOS-MIDDLE RIO GRANDE

Cyprinodon eximius (cachorrillo del Conchos, Conchos pupfish).—*Cyprinodon eximius* comprises at least four forms that may deserve taxonomic recognition (Miller, 1976). These include populations in the Río Conchos and Río Saúz (Chihuahua), a form in Devils River (Val Verde County, Texas), and a form in Río Grande tributaries upstream of Devils River (Presidio and Brewster counties, Texas, and Chihuahua). A protein electrophoretic survey provided genetic support for the distinctiveness of the Devils River population from other populations of the species (the Río Saúz population was not examined). The Devils River population was fixed for unique alleles at three of 30 loci examined; a population from Alamito Creek, a more upstream tributary of the Río Grande, was allozymically similar to two samples from the Río Conchos (Echelle and Echelle, 1998). An undescribed pupfish in Ojo de Villa López, an isolated spring near the Río Florido (Río Conchos drainage), may deserve species-level recognition (Contreras-B, 1991), but in overall appearance it resembles *C. eximius* and is genetically similar to that species (Echelle and Echelle, 1998).

The various forms of *C. eximius* occupy a variety of habitats ranging from constant temperature springs to eurythermal marshes and riverine situations.

The riverine forms can occur in relatively diverse assemblages of native fishes. For example, samples taken in 1901 from the Río Chihuahua at Chihuahua City, and from the Río Conchos at Camargo, Chihuahua, Mexico produced 12 and 16 species, respectively (Contreras-B, 1977), and, in 1994-1995, the species was taken with 8 to 19 species at four sites in the Río Conchos Basin (Edwards et al., 2001; Edwards et al., this volume). Although general collections from a site can produce a diversity of fishes, the pupfish tends to occupy shallow, quiet waters with relatively few species (Davis, 1980; Valdes-Cantu and Winemiller, 1997). Possibly because of differences in fish assemblage complexity, the species is rare in the mainstem of the Río Grande and more common in tributaries. In a series of 18 collections downstream of the mouth of the Río Conchos, Hubbs et al. (1977) found only a single specimen in the mainstem and this was at a site just downstream of Alamito Creek, where the species is more abundant.

In a 1994-1995 survey of 11 localities in the Río Conchos Basin, Edwards et al. (2001) found *C. eximius* abundant at a site in the Río Chuviscar and present, but less abundant, at one location each in the Río Conchos and two tributaries, Río San Pedro and Río Santa Isabela. The Río Florido was dry at sites visited

during that survey, whereas, in 1989, a collection of *C. eximius* was taken from that river near Villa López (Oklahoma State University Collection of Vertebrates, catalog number 18240). Populations associated with tributaries of the Río Grande between the Río Conchos and Devils River are restricted primarily to the downstream termini of small streams, although specimens are occasionally taken from associated waters in the Río Grande (Hubbs et al., 1977). The tributary streams are particularly vulnerable to habitat loss from dewatering as a result of upstream impoundments or pumping of groundwater. Construction of Amistad Reservoir in the 1960s inundated most sites of historic occurrence for the population in Devils River, and it was considered extirpated until its rediscovery in an 11-km reach of Devils River at the headwaters of the reservoir (Davis, 1980). In 1979, specimens from this area were transported to Dolan Creek in a successful effort to re-establish the population in Dolan Springs, a small (< 1 km) springfed area of historic occurrence in the Devils River drainage approximately 25 km upstream of the reservoir (Hubbs and Garrett, 1990).

The population of *C. eximius* in the Río Saúz Basin once was considered extinct because of drying of the habitat (Miller, 1961), but it was collected in 1964 from an impounded, possibly springfed, pond 75 km S of El Sueco, Chihuahua (Minckley and Koehn, 1965). In 1968 and 1975 it was collected in Estación Saúz and springs near Laguna Encinillas, but, in 1995 the former locality was dry, as was most of the surrounding area (SCB, unpubl.). The present status of this population is unknown.

Cyprinodon macrolepis (cachorrito escamudo, largescale pupfish).—This species is endemic to a rather large springfed pool, Ojo de la Hacienda Dolores, and its outflow, 12.5 km S-SW of Jiménez, Chihuahua, Mexico. Miller (1976) noted that the outflow “no doubt” once connected with the Río Florido, which supports *C. eximius*. Now the outflow has been highly modified into a number of irrigation distributaries and is isolated from the river. There is evidence from protein electrophoresis for past hybridization and genetic interaction between *C. macrolepis* and the population of *C. eximius* in the Río Florido (Echelle and Echelle, 1998), but the two species have maintained marked differences in color pattern and morphology (Miller, 1976).

During our visit in 1989, the species was common in the spring and the outflow immediately downstream of the headpool. The spring headpool is a locally popular recreation area and has been modified for swimming, with the edges shored up by rock walls. We are unaware of any analysis of the trend of the hydrograph for the springs.

Cyprinodon pachycephalus (cachorrito cabezón, bighead pupfish).—Two populations of “big-headed” pupfish occur in separate thermal-spring systems of the Río Conchos Basin (Minckley and Minckley, 1986): one described as *C. pachycephalus* in Baños (= Ojo) de San Diego, a small springfed system tributary to the Río Chuviscar, 57 km E of Ciudad Chihuahua, Chihuahua, Mexico, and an undescribed population in a spring near the Río Conchos at Julimes, 22 km SSE of Baños de San Diego. The taxonomic status of the latter population has not been determined, but they may represent the same species (Minckley and Minckley, 1986).

Baños de San Diego was described by Smith and Chernoff (1981) and Minckley and Minckley (1986). It comprises a small system of thermal springs that emerges on a small hilltop, and, prior to human alterations, must have emptied into the nearby Río Chuviscar (Minckley and Minckley, 1986). The system includes several small, commercially operated swimming pools and baths. Outlets from the springheads coalesce to form a small, partially braided, stream channel about 2 m wide and less than 3 cm deep. The spring run has been widened in places to form bathing pools, and, in 1971, it emptied into an artificial pond adjacent to the Río Chuviscar. However, by 1980 the spring outflow was diverted into a canal for irrigation (Smith and Chernoff, 1981). Water temperature is 43.8°C to 44.0°C in the springheads and cools as it moves downstream. The pupfish occurs throughout the springfed system alongside a possibly undescribed species (Contreras-B, 1991) related to the blotched gambusia (*Gambusia senilis*). Gonadal condition and other observations led Smith and Chernoff (1981) to conclude that *C. pachycephalus* and the local form of *Gambusia* are not stressed by the high temperatures and that the two exhibit “the highest long-term temperature tolerances known for any teleost.”

The only other fishes reported from the Baños de San Diego system are the more widespread pupfish *C. eximius*, a few putative *C. eximius* x *C. pachycephalus* hybrids, and longear sunfish *Lepomis megalotis*, all of which were restricted primarily to the tailwater pond in 1971 (Minckley and Minckley, 1981); none of these were reported by Smith and Chernoff (1981) from their visit in 1980. However, collections from the tailwater pond by one of us (SCB) in 1982 and by his student, Héctor Leal Sotelo, in 1984 included 16 species, including two non-natives, *Cyprinus carpio* and *Ameiurus melas*.

The status of this species is precarious because of its restricted habitat and the general trend toward loss of springflows as a result of over-pumping of groundwater in arid regions of the southwest (Contreras-B and Lozano-V, 1994). Human modifications of the habitat did not seem to threaten the species in 1989, when AAE and AFE visited Baños de San Diego. However, the present status of the species and the undescribed Julimes population is not well known.

TULAROSA BASIN

Cyprinodon tularosa (White Sands pupfish).—*Cyprinodon tularosa* is restricted to the Tularosa Basin in New Mexico where it occupies four isolated bodies of water, three on the White Sands Missile Range (Malpais Spring, Mound Spring, and Salt Creek) and one on Holloman Air Force Base near Alamogordo, Otero County. The Mound Spring and Lost River populations both represent translocations by local people in the late 1960s and early 1970s (Pittenger and Springer, 1999). Genetic analyses indicate that Salt Creek was the source for both introductions (Stockwell et al., 1998). Pittenger and Springer (1999) determined from historical records that Malpais Spring and Salt Creek have been modified by human activities, but Springer (FWS, pers. comm.) suggests that this might not have been associated with notable loss of habitat for the pupfish. The present, sharply incised nature of upper Salt Creek apparently occurred as a result of over-grazing and gully erosion sometime after the late 1800s (Pittenger and Springer, 1999; Springer, FWS, pers. comm.).

Habitats of *C. tularosa* show a wide range in salinity (1.5-60 ppt), temperature (3.0°C to 33.4°C) and environmental stability (Stockwell and Mulvey, 1998). Stockwell and Mulvey (1998) demonstrated a correlation between salinity and genotype at an allozyme locus that appears to reflect adaptation to salinity differences among habitats. In addition, parasitic infection in the White Sands pupfish is apparently a function of salinity (Stockwell et al., 1998). *Physa*, the intermediate host of diplostome trematodes, cannot tolerate salinities above 9 ppt. As a result, pupfish in

the relatively fresh Malpais Spring were highly infested (up to 100%) with white grub (*Posthodiplostomum minimum*), whereas the pupfish in Salt Creek were free of this parasite.

No other fish species occur in habitats supporting *C. tularosa* (Miller and Echelle, 1975). However, isolated waters in the vicinity of present populations do support introduced species: western mosquitofish (*Gambusia affinis*) at three sites, largemouth bass (*Micropterus salmoides*) at two, and goldfish (*Carrasius auratus*) at one (Pittenger and Springer, 1999).

Cyprinodon tularosa is considered a federal species of concern (Federal Register, 50:64481). Malpais Spring (266-ha wetland), Salt Creek (33-km stream) and Lost River (5-km stream) all support substantial pupfish populations, whereas the population in Mound Spring (two small ponds) is relatively small (Pittenger and Springer, 1999). Threats include introduction of exotic species, dewatering, and pollution, as well as disruption of the habitat by feral horses and off-road vehicle use. A conservation team organized in 1994 recommended establishment of additional populations in the Tularosa Basin. Stockwell et al. (1998) recommended that the Malpais Spring and Salt Creek forms be treated as separate units of conservation because loss of either one would result in a marked decrease in genetic diversity of the species. As previously mentioned, the Salt Creek form is represented in Mound Spring and Lost River. The Malpais Spring population, however, apparently has not been replicated.

LAGUNA DE GUZMÁN BASIN

Cyprinodon fontinalis (cachorrillo de Carbonera).—*Cyprinodon fontinalis* is known only from a series of five springs and their outflows near Ejido Rancho Nuevo in the Bolsón de los Muertos of the Guzmán Basin in northwestern Chihuahua. The springs are described in some detail and mapped by Smith and Miller (1980, 1981). They are separated by a maximum distance of only about 5 km. Ojo de Carbonera, the only spring habitat relatively unmodified by humans, includes a complex of spring-heads and flows about 100 m (less than 10 cm deep) before entering irrigation canals. In this system, *C. fontinalis* occurs most abundantly in small solution holes and along undercut banks of the spring outflow. The species also occurs in irrigation ditches and in four impounded springs. The only native fish sympatric with *C. fontinalis* is the largemouth shiner (*Cyprinella bocagrande*), which is known only from Ojo Solo (Chernoff and Miller, 1982), one of the five major springs supporting the pupfish. Black bullhead (*Ameiurus melas*) and western mosquitofish (*Gambusia affinis*) have been introduced into the area (Smith and Miller, 1980).

The species probably declined after human modifications to the habitat, including introductions of mosquitofish (*Gambusia affinis*), black bullhead (*Ameiurus melas*), and largemouth bass (*M. salmoides*), and construction of irrigation ditches and small impoundments of the springs (Smith and Miller, 1981). In the late 1970s, pupfish were more abundant in the relatively undisturbed Ojo de Carbonera than elsewhere in the area (Smith and Miller, 1980). The species was abundant at Ojo de Carbonera when we visited the area in 1989. An important threat is the possibility for increased pumping of groundwater for irrigation, a factor that has contributed to failure of springflows in nearby areas (Smith and Miller, 1981).

Cyprinodon albivelis (cachorrillo dorsal blanca, whitefin pupfish).—This species occurs in the western highlands of Chihuahua, primarily in the headwaters of the Río Papigóchic, a tributary of the Río Yaqui on the Pacific versant, and in a small springfed situation, Ojo de Arrey, in the Río Santa María sub-basin of the Laguna de Guzmán Basin. There is some question regarding whether the latter population is native or a result of a modern introduction from the Río

Papigóchic, but Minckley et al. (2002) suggested, on the basis of genetic considerations (Echelle and Dowling, 1992; Echelle and Echelle, 1993a) and gill raker counts, that it probably is a native population.

The species was widespread and abundant in the Río Papigóchic system in 1978 (Hendrickson et al., 1981) and in 1982 and 1986 (SCB and MLLV). In 1989, it was locally abundant at a site in the Río Papigóchic and at two different springs in the Ojo de Arrey system (AAE and AFE), and, according to Minckley et al. (2002), P. J. Unmack found the latter population “intact in 1999.” Minckley et al. (2002) noted that “. . . local stocks . . . [probably] have disappeared due to increased human activities, . . . [but they had] no reason to believe the species is as yet in jeopardy.”

Cyprinodon pisteri (cachorrillo de Guzmán, Guzman pupfish).—This species (= “Palomas pupfish” in some publications; e.g., Echelle and Echelle, 1998) occurs relatively widely in the Guzmán Basin, and apparently introduced populations once occurred just across the U.S./Mexico boundary in New Mexico (Minckley et al., 2002). The Lago de Guzmán complex comprises the ríos Casas Grandes, Santa María, del Carmen (= Río Santa Clara), and the Laguna Bustillos Basin. In March 1990, Propst and Stefferud (1994) surveyed these basins for the occurrence of Chihuahua chub (*Gila nigrescens*), and collected the pupfish at one or more sites in each, except that the species was absent from their nine sample sites in the Río Santa Clara. Minckley et al. (2002) noted that “the species is catholic in habitat, occupying springs, marshes (ciénegas), shorelines and cutoff channels of rivers and creeks, even colonizing ephemeral canals and ditches along roadsides.”

Anthropogenic factors such as habitat destruction, degradation and fragmentation, pollution, and nonnative species have adversely affected native aquatic communities over a large portion of the Guzmán Basin (Propst and Stefferud, 1994). Minckley et al. (2002) observed that, although the Guzmán pupfish is relatively widespread, losses have occurred with the drying of a diversity of aquatic habitats at lower elevations in the Guzmán Basin. They cite Brand (1937) as follows: “The increasing use of spring and river water for irrigation in the *haciendas* and *colonias*

of the region has contributed markedly to the lessened flow of the rivers in their lower courses. Many of the abundant springs that fed this system ~80 years ago have failed." However, some losses of habitat, includ-

ing, by about 1975, the type locality, a springfed ciénega near Las Palomas, are attributable to over-pumping of groundwater on both sides of the international boundary (Minckley et al., 2002).

CUATRO CIÉNEGAS

Cyprinodon atrorus (cachorrito del Bolsón, bandfin pupfish).—*Cyprinodon atrorus* occurs primarily in physicochemically variable, often ephemeral habitats in the Bolsón de Cuatro Ciénegas de Carranza (Cuatro Ciénegas Basin), Coahuila, México. However, the species also occurs in stable, springhead environments in the southeastern end of the Cuatro Ciénegas Basin, where the other pupfish of the basin (*C. bifasciatus*) is absent (Arnold, 1972). The two Cuatro Ciénegas pupfishes hybridize in areas where they come into contact either naturally or because of human-constructed irrigation canals (Miller, 1968; Minckley, 1977). However, such hybridization is restricted to local zones of contact and, outside these zones, the two species are maintaining their morphological and genetic distinctiveness (Echelle and Echelle, 1998; E. Carson, pers. comm.).

The species can be locally very abundant, but is considered rare because of its restricted distribution (Williams et al., 1989). The amount of habitat available for pupfish must have been much more extensive at the turn of the century, when Cuatro Ciénegas was a closed basin with no outflow (Rodríguez González, 1926, as cited by Calegari, 1997). Now, however, canals transport water outside the valley (Minckley, 1969, 1977). Canalization of springs and their outflows for agriculture and industry, mostly outside the basin (Contreras-B, 1991), undoubtedly has reduced the amount of habitat available to this fish. In November 1994, the federal government designated Cuatro Ciénegas as a National Protected Area, affording some protection for the system. The species remains abundant, despite a long history of human activity in the basin, including agriculture, gypsum mining, tourism,

and a recent increase in manufacturing plants (Calegari, 1997). Uncontrolled tourism and continued economic pressures on the human population pose a variety of threats (Contreras-B, 1991; Calegari, 1997), and there is some concern that groundwater pumping is threatening spring flows by causing the water table to fall (Grall, 1995). The ultimate impact of introduced species, including African cichlids (*Oreochromis* sp. and *Hemichromis guttatus*), water hyacinth (*Eichornia crassipes*), and Asian snail (*Melanoides tuberculata*), remains to be seen.

Cyprinodon bifasciatus (cachorrito de Cuatro Ciénegas, twoline pupfish).—*Cyprinodon bifasciatus* occupies an arc of constantly warm springs and their outlet pools and streams in the valley floor around the northern tip of Sierra de San Marcos, an area of karst topography where there are cenote-like, springfed sinkholes ("pozas") that range up to 200 m in diameter and more than 10 m deep (Miller, 1968; Minckley, 1969, 1977). Ecologically, the species is largely segregated from *C. atrorus*, the other pupfish of the Cuatro Ciénegas Basin, but they meet and hybridize in peripheral areas (see account for *C. atrorus*).

The species is considered vulnerable to extinction because of its restricted distribution and rather specialized habitat requirements (Williams et al., 1989). Some of the pozas have been developed for picnicking and swimming, but these activities do not seem to pose serious threats to the species. See the account for *C. atrorus* for additional comments on introductions of non-native species and other human activities in the basin.

DISCUSSION

None of the 12 pupfishes endemic to the northern Chihuahuan Desert, as defined for purposes of this symposium, has gone extinct. However, two spe-

cies, *C. bovinus* and *C. elegans* have declined in range by at least 50% and, without intensive management, native stocks of the former would have been lost to

introgressive hybridization. The latter factor has eliminated native stocks of *C. pecosensis* over about 80% of its historic range. With one major exception, most of the remaining species have undergone various degrees of range contraction as a result of a variety of anthropogenically induced losses and physical alterations of habitat. The exception is *C. tularosa*, which is confined within a military reservation with highly restricted access. The range of this species has expanded within its general area of endemism as a result of transplantations by local people (Pittenger and Springer, 1999) into previously unoccupied waters in the Tularosa Basin.

It is well understood that introduced non-native fishes are a major factor in the decline of native fishes in southwestern North America (Moyle et al., 1986; Allendorf and Leary, 1988; Propst et al., 1992) including northern Mexico (Contreras-B et al., 1976; Contreras-B and Escalante, 1994; Contreras-B, 2000). Correspondingly, depletion of pupfish populations via competition and predation by non-natives has been inferred for various situations in southwestern North America (Soltz and Naiman, 1978; Schoenherr, 1981). However, the potential for losses as a result of genetic introgression by non-native pupfish may have been underestimated until recently. The rapidity with which this factor can cause losses of native stocks is dramatically demonstrated by events following the introduction, in the 1960s (Stevenson and Buchanan, 1973), of *C. variegatus* into Lake Balmorhea. Apparently because of physical barriers, the locally endemic pupfish, *C. elegans*, has been little affected, but the presence of a dense, nearby population of the non-native poses a continual threat for this species (A. F. Echelle and Echelle, 1994) and other endemic pupfishes of the region. Genetic markers indicate that the Lake Balmorhea population of *C. variegatus* has been the source for subsequent introductions into both Diamond Y Draw (Echelle and Echelle, 1997) and the Pecos River (Childs et al., 1996) which support, respectively, the endemic species *C. bovinus* and *C. pecosensis*. As described in our accounts for the pupfishes of the Pecos River Basin, the resulting rate and extent of genetic introgression has generated a great deal of concern and costly, sometimes futile, management activity for the endemic pupfish species.

Smith (1981) argued convincingly that pupfishes in desert environments are hardy generalists ultimately derived from estuarine ancestors pre-adapted to sur-

vive the Post-Pleistocene desiccation of pluvial Pleistocene waters (Miller, 1981) of southwestern North America. After surviving and thriving during the post-Pleistocene expansion of desert, the pupfishes in the Chihuahuan Desert have been exposed to dramatically rapid anthropogenic reductions in habitable surface waters since the early 1800s. Since those years, arid grasslands were replaced by shrub desert over large portions of the desert southwest, an effect largely attributable to Anglo-American settlement, agriculture, and domesticated livestock (York and Dick-Peddie, 1969; Gehlbach, 1981; Hendrickson and Minckley, 1984). The rate of desertification, with losses of springs and ciénegas and reduced persistence of natural stream habitat, was hastened by the more recent advent of mechanized construction of dams, canals, and water diversions, and groundwater pumping.

Groundwater pumping may be the ultimate threat to most of the spring-dwelling pupfishes of the Chihuahuan Desert. In the past few decades, groundwater pumping has delivered the *coupe de grace* to many springs in the northern Chihuahuan Desert (Scudday 1977, this volume; Brune, 1981; Contreras-B and Lozano-V, 1994), and springs of the region continue to fail (Contreras-B and Lozano-V, 1994; Sharp et al., this volume). In southern Nuevo León, just south of the region treated in our species accounts, five species of *Cyprinodon* and the monotypic genus *Megupsilon*, the closest relative of *Cyprinodon* (Echelle and Echelle, 1993b; Parker and Kornfield, 1995), were driven to extinction in the wild within 15 years of the advent of intensive groundwater pumping (Contreras-B and Lozano-V, 1994, pers. observ.). In the northern Chihuahuan Desert, groundwater pumping seems directly responsible for losses of two of three morphologically divergent populations of *C. elegans* and significant losses of populations of *C. bovinus* and the Guzman pupfish. In addition, there almost certainly have been losses of populations that may have occupied now-dry springs whose original faunas were unknown to science (W. L. Minckley, pers. comm.; SCB and MLLV, unpubl. data).

Persistence of habitats suitable for the great variety of often locally endemic aquatic and semi-aquatic organisms in desert ecosystems (Williams et al., 1985) is especially threatened when drought conditions are superimposed onto a landscape already depleted of water resources by human activities. This probably explains the recent failure of Phantom Lake Spring

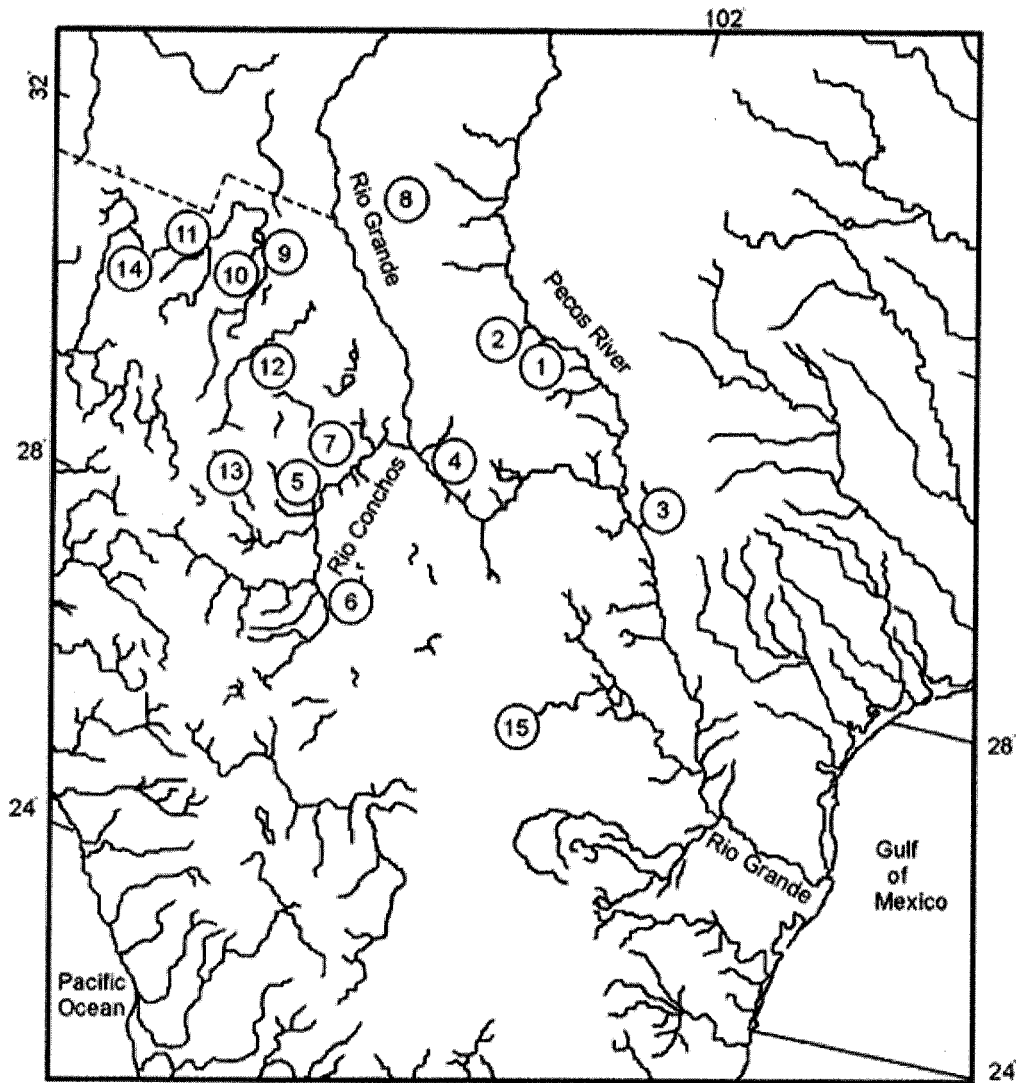


Figure 1. Basins and springs mentioned in the text. 1 = Diamond Y Draw, 2 = Toyah Creek, 3 = Devils River, 4 = Alamito Creek, 5 = Baños de San Diego, 6 = Ojo de la Hacienda Dolores, 7 = Río Saúz, 8 = Tularosa Basin, 9 - 13 = Laguna de Guzmán Basin, 9 = Bolson de los Muertos, 10 = Río Santa María, 11 = Río Casas Grandes, 12 = Río del Carmen, 13 = Laguna Bustillos, 14 = Río Papigóchic, 15 = Cuatro Ciénegas.

and the decline in surface waters in Diamond Y Draw, both of which are described earlier in this paper. It remains to be seen whether surface waters in these situations will rebound when the present drought ends. Regardless, with time and continued over-pumping of groundwater, these spring systems and others will fail as part of a trend of such failures in the past few de-

acades (Brune, 1975, 1981; Contreras-B and Lozano-V, 1994; Sharp et al., this volume). This can be avoided only if society decides that these aquatic ecosystems are worth preserving, perhaps for their own sake or for their aesthetic appeal or what they signal about water availability for future human needs.

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SPRING-ENDEMIC *GAMBUSIA* OF THE CHIHUAHUAN DESERT

CLARK HUBBS

ABSTRACT

Spring endemic fishes are restricted to the vicinity of spring outflows where stenothermal conditions prevail. Stream fishes occupy downstream locations that are substantially more eurythermal. Those circumstances prevail throughout the Chihuahuan Desert where *Gambusia senilis* interacts with *G. hurtadoi* and *G. alvarezi* and *Gambusia affinis* interacts with *G.*

nobilis and *G. gaigei*. *Gambusia amistadensis* once occurred in and was restricted to Goodenough Spring where it interacted with *G. affinis* in the Rio Grande. All evidence indicates that similar interactions occur with other fishes, amphipods, crayfish, and salamanders.

INTRODUCTION

The fishes of the Chihuahuan Desert require adequate quantity and quality of water. In as much as few lakes occur there, the need is for lentic environments. Throughout most of the southwest United States and northwest Mexico, human desires often negatively impact stream flows. Problems associated with all the fishes can be illustrated by problems with members of the genus *Gambusia*. Many of the fishes are spring endemics that require natural spring flow volumes (Hubbs, 1996, 2001). Often the problem is intrageneric. For example, *Gambusia senilis* and the *affinis* species group are stream fishes and *G. nobilis*, *G. gaigei*, *G. hurtadoi*, and *G. alvarezi* are spring endemics. When spring flow volumes decrease, the stream species gain area at the expense of the spring endemics. Stream species have a competitive advantage away from springs and spring species have a competitive advantage in springs. The primary driving factor is stenothermal vs. eurythermal conditions. Similar interactions occur within cyprinids, centrarchids, and percids in the Chihuahuan Desert. Likewise in Nevada *Crenichthys* is adapted to stenothermal, warm, low dissolved oxygen springs and many native minnows are better as stream fishes. *Crenichthys* can survive in below 0.7 ppm O₂ at 37°C (Sumner and

Sargent, 1940; Hubbs and Hettler, 1964). It is unlikely that members of the genus *Cyprinodon* have similar needs for stenothermal water, as *Cyprinodon elegans* populations increased at Phantom Cave Spring when spring flow declined (Hubbs, 2001). Nevertheless, *Cyprinodon* clearly needs adequate volumes of water even if it is not stenothermal.

The needs of spring fishes for quantities of spring flow has also been reported from Alabama (Howell and Black, 1976), Arkansas (Robison and Buchanan, 1988), New Mexico (Hubbs and Echelle, 1972), Oklahoma (Matthews et al., 1985), China, France, Italy, Croatia (Maurice Kottetot, European Ichthyological Congress, pers. comm.), Australia, Brazil, Congo, and Iran. I consider spring fish to be defined as species found wholly or most often in spring heads and/or in the immediate zone downstream (i.e., spring runs) and seldom found elsewhere in the drainage. Many other aquatic organisms have similar spring vs. stream abundances including amphipods (Bowles and Arsuffi, 1993), aquatic plants (Emery, 1967) salamanders (Tupa and Davis, 1976; Chippendale et al., 1993) and crayfish (Hubbs, 2001). I suspect the entire aquatic biota is also involved but commonly not as well known.

EXAMPLES IN *GAMBUSIA*

Gambusia nobilis.—The Pecos gambusia is a relatively robust *Gambusia*, with an arched back and a caudal peduncle depth that is approximately two-thirds of the head length. The margins of the scale pockets are outlined in black and spots are normally absent on the caudal fin, however, sometimes a faint medial row of spots may be present. The dorsal fin has a sub-basal row of spots. Females have a prominent black area on the abdomen that surrounds the anus and anal fin. The male gonopodium has a number of unique features including elongated spines on ray 3, small rounded hooks on the tips of rays 4p and 5a, and an elbow on ray 4a consisting of 3 or 4 fused segments located opposite the serrae of ray 4p. *Gambusia nobilis* generally have 8 dorsal, 12 pectoral, 9 to 10 anal, and 6 pelvic rays (Hubbs and Springer, 1957; Koster, 1957; Bednarz, 1975; Echelle and Echelle, 1986). Populations in Toyah Creek (Texas) and Blue Spring (New Mexico) were found to be the most diverse morphologically and genetically and the Toyah Creek population had the greatest genetic heterogeneity (Echelle and Echelle, 1986; Echelle et al., 1989). *Gambusia nobilis* was described by Baird and Girard (1853) based on material from Leon and Comanche springs, Pecos County, Texas. Leon Springs was later designated the type locality (Hubbs and Springer, 1957). The species is endemic to the Pecos River basin in southeastern New Mexico and western Texas and originally ranged from near Fort Sumner, New Mexico to the area around Fort Stockton, Texas. At present, the species is restricted to four main areas, two in New Mexico and two in Texas. Populations live in various springs and sinkholes in Bitter Lake National Wildlife Refuge, near Roswell, New Mexico; Blue Spring, east of Carlsbad Caverns National Park, New Mexico; the Diamond Y springs and draw (= Leon Creek), near Fort Stockton, Texas; and the Balmorhea springs complex and Toyah Creek near Balmorhea, Texas. Extirpated populations include the Pecos River near Fort Sumner and North Spring River in New Mexico, and Leon and Comanche springs, in Texas. Those populations in Comanche Springs were extirpated when the spring dried (Hubbs and Springer, 1957). The population in Leon Springs was eliminated by impoundment (Miller et al., 1991). Those populations in the Balmorhea springs complex and the Diamond Y region are most abundant in stenothermal waters.

Where suitable habitats exist, Pecos gambusia populations can be dense. An estimated 27,000 individuals inhabit the Bitter Lake National Wildlife Refuge area, and 900,000 inhabit Blue Spring (Bednarz, 1975, 1979). Approximately 100,000 Pecos gambusia are estimated to inhabit the Balmorhea springs complex and more than 100,000 in the Diamond Y springs and draw. Pecos gambusia primarily inhabit stenothermal springs, runs, spring-influenced marshes (ciénegas), and irrigation canals carrying spring waters. Some populations however, are also known from areas with little spring influence; these habitats generally have abundant overhead cover, and include sedge covered marshes and gypsum sinkholes (Echelle and Echelle, 1980). One or two other *Gambusia* may also be found in association with *G. nobilis*. Where the western mosquitofish (*G. affinis*) is found, *G. nobilis* inhabits stenothermal waters and *G. affinis* is most often found in eurythermal habitats. Where together, the Pecos gambusia is much more likely to be found associated with vegetation or in deeper waters, while *G. geiseri* tends to be at the surface or in open water over non-vegetated substrates (Hubbs et al., 1995). Another spring endemic, *G. geiseri* was introduced into Pecos gambusia habitat for "mosquito control." This was a serious error as the native *G. nobilis* is at least as good at mosquito control. Pecos gambusia feed relatively non-selectively, consuming a diversity of food types, including amphipods, dipterans, cladocerans, filamentous algae, arachnids and mollusks (Hubbs et al., 1978; Winemiller and Anderson, 1997). *Gambusia nobilis* produce live young. Bednarz (1979) reported that the number of embryos was related to female size and that the mean number of embryos was 38 in the Blue Spring population. Hubbs (1996) found that the birth weight of Pecos gambusia from Texas populations ranged between 35 and 50 mg and females had an interbrood interval of 52 days. Hybrids between *G. nobilis* and *G. affinis* or *G. geiseri* are occasionally found, especially in habitats where one of the species is rare (Hubbs and Springer, 1957).

Pecos gambusia face severe threats from spring flow declines and habitat modification throughout their range. In parts of their range, ciénegas, presumed to have supported large numbers of *G. nobilis*, were drained and spring flows diverted for irrigation. San

Solomon Springs has been modified into a spring-fed swimming pool. During 1998 and 1999, Phantom Lake (part of the Balmorhea springs complex) dried twice and after the second drying, Phantom Lake Spring ceased to flow and ultimately the Phantom Lake Spring refuge canal dried. During May 2000, a pump was installed by the U.S. Bureau of Reclamation to insure continuous flow in the upper spring pool. Additional stresses on the population may occur through competition with the introduced *G. geiseri*. Efforts have been made to improve habitat in the Balmorhea area. A small refugium canal was constructed in 1974 in Balmorhea State Park (Echelle and Hubbs, 1978). In 1993, the Bureau of Reclamation constructed a modified 110-m canal at Phantom Lake Spring (Young et al., 1994) with sloped, sinuous sides to resemble a portion of a ciénega. That canal is now dry and all fish dead as Phantom Lake Spring no longer flows. After Phantom Lake Spring ceased flow and the only water remaining was in the "spring pool" that is artificially maintained by a pump, *G. nobilis* declined precipitously, *G. affinis* increased, and *G. geiseri* remained rather stable (Hubbs, 2001). In cooperation with local residents and farmers, in 1996 the construction of the 1-ha San Solomon Ciénega was completed (McCorkle et al., 1998; Garrett, this volume). This wetland is situated within the boundaries of the original, natural ciénega on state park land. Designed to resemble and function like the original ciénega, the native fish fauna, including *G. nobilis*, has flourished near the inlet where stenothermal conditions prevail. The park refugium canal, and the San Solomon Ciénega have increased numbers and security for the species, but each remains dependent on spring flows.

The Diamond Y introductions of *G. geiseri* were eliminated by efforts to remove exotic *Cyprinodon variegatus* that formed massive hybrid swarms with the endemic *C. bovinus* that threatened the survival of the latter (Hubbs et al., 1978). A byproduct of those control efforts was the eradication of exotic *G. geiseri* from the Diamond Y system. Equivalent collecting effort at Diamond Y Spring before eradication had equal numbers of *G. geiseri* and *G. nobilis*. After eradication, *G. nobilis* had abundances similar to the sum of the two species abundances. Where *G. geiseri* was absent *G. nobilis* dominated stenothermal water and *G. affinis* dominated eurythermal waters. In the Balmorhea area three species are involved. In general the 2 spring species dominate the stenothermal water and *G. affinis* dominates in the eurythermal waters.

There is a strong tendency for *G. nobilis* to be restricted to the most stenothermal waters, and *G. geiseri* to be common in waters of intermediate thermal variation. Additional protection for *G. nobilis* stems from its presence in Balmorhea State Park, Bitter Lake National Wildlife Refuge, Diamond Y Draw which is owned by The Nature Conservancy of Texas, and an introduced stock of Pecos gambusia in artificial pools at the Living Desert State Park near Carlsbad, New Mexico.

Gambusia gaigei.—The color pattern of the Big Bend gambusia is weaker than that of the other members of the *G. nobilis* species group. The ground color is silvery with an iridescent blue overtone. There is considerable yellowish-orange on the clear areas of the unpaired fins. The markings on the margins of the scale pockets are faint. There are none anterior to the anus or below the eye. The middorsal streak and dark coloration of the neurocranium cover and obscure the scale-pocket markings, but the postanal streak does not obscure these markings. The faint, broad lateral band often obscures the scale-pocket markings on one scale row. There are a few dark crescents on the scale rows surrounding the lateral band. The anal spot in females is restricted to the anus. The dorsal has a subbasal row of black spots and a dark margin. The caudal has no dark markings. The anal of females has a dark margin; that of males is grayish. There is only a trace of a dark chin bar. Except for the suborbital bar there is no darkening of the lower parts posterior to the eye.

The Big Bend gambusia was described in Hubbs (1929) based on specimens collected by F. M. Gaige from "a marshy cattail slough fed by springs, located close to the Rio Grande at Boquillas, Brewster County, opposite the Mexican village of the same name." Apparently this was the largest river-side spring in the Big Bend region (Hubbs, 1940). Subsequently in June 1954 numerous specimens were obtained from Graham Ranch Warm Springs (Hubbs and Springer, 1957). Graham Ranch Warm Springs (now known as Spring 4) is the largest spring near Boquillas but is 1 km west of Boquillas. Other springs existed at Boquillas that may have been the source of Fred Gaige's captures. Those springs dried in 1954 and no longer contain fish. It is more likely, however, that the original collections came from Graham Ranch Warm Springs.

Although Big Bend gambusia were abundant in Graham Ranch Warm Springs and the newly constructed adjacent "kiddie fishing pool", they were scarce by 1956 and the previously scarce *G. affinis* very abundant. Consequently, renovation efforts were initiated 9 October 1956. Intensive seining obtained 24 individuals and the area was treated with rotenone and fewer than 12 other Big Bend gambusia (and thousands of *G. affinis*) killed (Hubbs and Broderick, 1963). The 24 remaining individuals were placed in 5 locations: 1) Boquillas Spring, Glenn Springs and, stock tank along the Glenn Springs Road (14 individuals); 2) metal tank near the Park Headquarters (6 individuals) and; 3) 4 individuals taken to Austin. The fish placed at location 1 were never seen again. The fish at location 2 flourished until a cold day that killed them all. One of the 4 remaining fish died in Austin but the other 3 (1 female and 2 males) were returned to the park in a newly constructed pond where they flourished (Hubbs and Broderick, 1963). At the same time the Rio Grande Village Camp Ground was established near the Graham Ranch Warm Spring (and the existing 4 springs renamed 1-4). Trees were planted to shade the camp ground and watered from the Rio Grande. That water drained into the Big Bend gambusia pond. *Gambusia affinis* got into those irrigation ditches and subsequently into the Big Bend gambusia pond.

When the pond was examined 16 April 1960 only 27 individuals were obtained. All were taken to Austin. Half of them were sent to the University of Michigan for insurance against extinction. Both cultures flourished, and after a second refuge pond (using Spring 1 water) was constructed, were returned (9 August 1966) to the park (Hubbs and Williams, 1976). With 2 intervals of extreme scarcity, it is not surprising that the surviving population is homozygous (Echelle and Echelle, 1991). Minor problems (introduction of green sunfish, *Lepomis cyanellus*, and minor mortality due to an extremely cold day) did not threaten survival (Hubbs and Williams, 1976). Subsequently all Spring 4 water was diverted to the Campground and the kiddie fishing pool dried. Later a flash flood ran through the Spring 1 refuge pool into the Spring 4 pool, reintroducing the Big Bend gambusia into its original habitat. Increased pumpage caused the Spring 1 refuge pool to overflow through its drain pipe. Fish were transferred from the refuge pool into

the overflow ditch and both flourish. Eventually, water leaked from the Spring 4 pool and some Big Bend gambusia occurred in a large beaver pond between Spring 4 and the Rio Grande. Big Bend gambusia flourishes in the 2 spring pools and the Spring 1 drainage ditch. A small population occurs in the Beaver Pond. A small number of *G. affinis* occurs in the Spring 4 pool. The relative number of *G. gaigei* and *G. affinis* is correlated with the thermal stability of the two localities. *G. gaigei* dominates in stenothermal water and *G. affinis* dominates in the eurythermal water (Hubbs, 2001).

In Chihuahua, Mexico a similar pattern emerges but much less data on environmental conditions are available.

Gambusia hurtadoi.—The Dolores gambusia lives in El Ojo de la Hacienda Dolores that has a large spring pool ca 20 X 50 m with a dense population of *G. hurtadoi* as well as an endemic *Cyprinodon*. Both endemics also flourish in the irrigation ditches leaving the spring pool. During wet weather, spring waters empty into the Rio Florido where *G. senilis* is the only gambusia present.

The color pattern of the Dolores gambusia is darker and has more iridescent blue than that of other members of the *G. nobilis* species group. The ground color is iridescent bluish-silver. There is less orange on the body than in *G. alvarezi*. The clear areas on the median fins are yellowish orange. The markings on the margins of the scales are dark but diffuse, and are often obscured by other more prominent markings. There are no marks anterior to the anus and none below the preopercle. The lateral diffusion of the dark middorsal streak often reaches the lateral band. The thin postanal streak does not obscure the scale-pocket markings. The lateral band is broad and dark. The dark crescents on the side are concentrated along the lateral band and often cover and obscure it. They follow the scale-pocket margins. The anal spot is not restricted to the anus. The dorsal fin has a subbasal row of spots, which are darker than the darkened margin. The caudal has no dark markings. The anal fin of females has a dark margin; that of males is grayish. The prominent dark chin bar is often interrupted medially. The suborbital bar is dark.

Three color variants have been noted. These may be designated Spotted, Gray, and Golden. All three lack the dorsal streak, the post-anal streak, and the lateral band. Spotted also has no dark markings on the scale-pockets and no suborbital bar; the color pattern on the body consists chiefly of the crescents which are often grouped and not concentrated along the location of the lateral band. Gray lacks the crescents typical of Spotted. Its coloration somewhat resembles that of *G. geiseri* except that the scale-pocket markings are more diffuse. Golden has no dark markings except the dark margin of the dorsal, three small patches of scale-pocket margins in front of the dorsal and above the pectoral bases, the spotting of the peritoneum, and the dark eye. In life it is an attractive golden yellow. There is no iridescent blue. All three color phases were present in a collection made by Clark Hubbs and Oscar f. Wiegand at El Ojo de la Hacienda Dolores on December 31, 1954. The preserved collection contained a random sample of 1,342 individuals. Later a single Golden individual 16 mm long was collected. Approximately 500 more specimens were collected in the subsequent work. Sixteen Spotted and thirteen Grays have been noted in the preserved sample. Although 502 of the specimens are longer than 20 mm, none of the color variants are. It is possible that they have a reduced survival rate. This hypothesis is supported by our failure to bring the Golden individual back to Austin alive.

Gambusia alvarezi.—The color of the San Gregorio gambusia is more yellow and orange than that of any other members of the *G. nobilis* species group. The ground color is yellow-orange. There is little iridescent blue on the body. All clear areas on the fins are yellow-orange. The diffuse dark markings on the margins of the scales are often obscured by more prominent markings. There are no marks anterior to the anus or below the eye. The middorsal streak is dark, but its lateral diffusion does not reach to the lateral band. The dim postanal streak often obscures the scale-pocket markings. The lateral band is broad and dark. The dark crescentic marks on the side are concentrated along the lateral band. They are not numerous and more than three are seldom interconnected. The lateral band can easily be traced through the areas between the scattered groups of crescents. The black spot around the anus is large. The dorsal fin has a subbasal row of spots and a darker margin. The caudal has no dark markings. The anal fin of females has

a dark margin; that of males is uniformly grayish. The prominent dark chin bar is often interrupted medially. The suborbital bar is dark.

Similarly, El Ojo de San Gregorio has a dense population of *G. alvarezi*. El Ojo de San Gregorio has much less volume than El Ojo de la Hacienda Dolores and virtually no spring pool. The water eventually flows (or flowed) into the Rio Parral once occupied by *G. senilis*. Unfortunately the Rio Parral is severely polluted with mining wastes that preclude contact between the two species.

Color differences were maintained in laboratory-reared stocks of *G. alvarezi*, *G. gaigei*, and *G. hurtadoi*.

Gambusia amistadensis.—The Amistad gambusia was a member of the *Gambusia senilis* species group and is closely related to *G. hurtadoi*, *G. alvarezi*, *G. gaigei*, and *G. senilis* (Rauchenberger, 1989). The species is characterized by its relatively slender body, terminal mouth with numerous teeth on each jaw, and males having long serrae on ray 4p of the gonopodium. Preserved specimens have strong crosshatching and numerous darkly pigmented crescent-shaped spots on their scale margins. The mid-dorsal stripe is narrow and the lateral stripe is broad. A short, dusky subocular bar is present. Adult females have a permanent median dark anal spot (Peden, 1973).

The Amistad gambusia was originally described from Goodenough Springs (29°32'10"N, 10°15'10"W) in Val Verde County, Texas. The original range of the species included the headsprings and the 1.3-km spring run downstream to its confluence with the Rio Grande (Peden, 1973). The species became extinct in the wild when Goodenough Springs, once the third largest spring system in Texas, was inundated by Amistad Reservoir when the dam gates were closed in 1968 (Peden, 1973; Brune, 1981). Goodenough Springs and its warm spring run rapidly flowed over limestone gravel and sand substrates along its course to the Rio Grande. Waters originated in the relatively large Edwards-Trinity aquifer (Peckham, 1963) and maintained flow rates of approximately 2,000-4,000 cubic liters per second (Brune, 1981). The type locality and habitat for the Amistad gambusia is now under approximately 30 m of water and former spring openings may now be recharge zones (Peden, 1973; Brune, 1981).

Little is known concerning the food habits of the Amistad gambusia; however, the gut contents of 10 paratypes examined by Peden (1973) contained mostly unidentified items, some insect fragments and traces of filamentous algae. Other fishes co-occurring with the Amistad gambusia prior to the inundation of its habitat included: *Astyanax mexicanus* (Mexican tetra), *Macrhybopsis aestivalis* (speckled dace), *Cyprinella lutrensis* (red shiner), *C. venusta* (blacktail shiner), *C. proserpina* (proserpine shiner), *N. braytoni* (Tamaulipas shiner), *N. jemezianus* (Rio Grande shiner), *Cycleptus elongatus* (blue sucker), *Ictalurus punctatus* (channel catfish), *Ameiurus melas* (black bullhead), *Pylodictis olivaris* (flathead catfish), *Gambusia affinis* (western mosquitofish), *Micropterus salmoides* (largemouth bass), *Lepomis cyanellus* (green sunfish) and *Cichlasoma cyanoguttatum* (Rio Grande cichlid) (Peden, 1973).

Observations in aquaria by Peden (1970, 1973) indicated that male courtship appeared similar to that

found in other poeciliids and that pregnant female *Gambusia amistadensis* gave birth to their young in vegetated areas. Of 10 female paratypes examined by Peden (1973), the mean size was 29.8 mm SL (range = 25.9-34.6 mm SL) and 7 contained 5 to 11 (mean 8.9) embryos in each ovary while the other 3 females contained 1 to 7 eggs.

Culture populations of *G. amistadensis* were maintained until the late 1970s at the University of Texas at Austin and at the U.S. Fish and Wildlife Service's endangered species culture facility in Dexter, New Mexico (Hubbs and Jensen, 1984). These populations were contaminated by western mosquitofish (*Gambusia affinis*), which eliminated the *G. amistadensis* in these cultures prior to 1983 (Hubbs and Jensen, 1984). I suspect this species would not be extinct if those refugia had been maintained as stenothermal environments.

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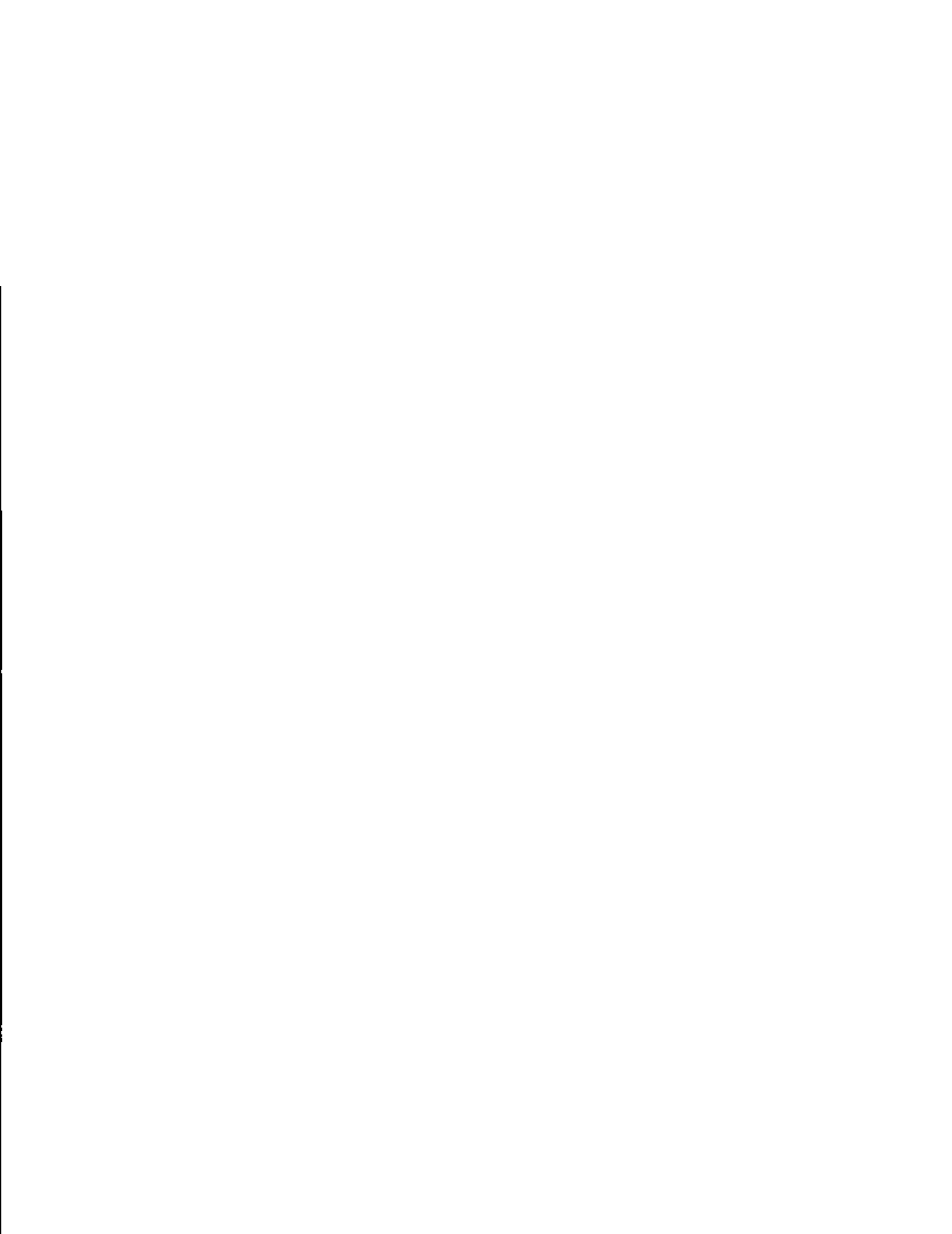
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MY FAVORITE OLD FISHING HOLES IN WEST TEXAS: WHERE DID THEY GO?

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ABSTRACT

I provide a historical perspective of water resources that were present in Trans-Pecos Texas, particularly Pecos County, from 1940-1980, with notes on the fish and other aquatic organisms that existed in these springs, creeks, lakes and rivers. Recreational fishing for game fish was the impetus for visiting these fishing holes. This in turn led to a young man developing an interest in the natural history of the region. By the late 1950s, the springs were nearly all gone,

wetland marshes disappeared, creeks disappeared, and the rivers looked more like sluggish creeks. The drought of the 1950s, coupled with the boom in pump irrigation projects, spelled the doom for this aquatic waterland. The current drought, in combination with ongoing water development projects, has further degraded the aquatic environment. The future existence of native Chihuahuan Desert fishes under the current trends is unlikely.

COMANCHE SPRINGS

I was fortunate as a youth growing up in Ft. Stockton, Texas, to be able to do so in a somewhat Tom Sawyerish way. My father was an avid outdoorsman, and he often took my brother and me with him on his many fishing excursions throughout west Texas. Fort Stockton was the center of an astonishing number of fishing holes in an otherwise desert environment. Natural springs gushed forth from numerous locales in all directions from Fort Stockton. The largest series of springs were right in the town of Fort Stockton. The entire Comanche Springs complex consisted of numerous springs along a nearly half-mile stretch of an arroyo bordering the east side of town. The largest single spring was the Chief Spring, a large spring that flowed an average of 35 million gallons (132 million liters) per day. The combined flow of the Chief with the other lesser springs amounted to an outflow of 66 million gallons (250 million liters) per day. This water was tapped early in the history of the area for irrigation of croplands north and northeast of town. As much as 6,200 acres (2,509 ha) of land was eventually watered from Comanche Springs (Brune, 1975).

A municipal swimming pool was constructed around the Chief Spring during the 1930s, and a city park developed around the arroyo bottom to encom-

pass all the other free-flowing springs. Canals from all these springs converged into one large canal just below the swimming pool. Native species of fish such as Mexican tetra (*Astyanax mexicanus*), Comanche Springs pupfish (*Cyprinodon elegans*), mosquitofish (*Gambusia affinis* and/or perhaps *G. nobilis*), roundnose minnow (*Dionda episcopa*) and channel catfish (*Ictalurus punctatus*), along with the common snapping turtle (*Chelydra serpentina*) and the yellow mud turtle (*Kinosternon flavescens*), co-existed in this abundance of water. Largemouth bass (*Micropterus salmoides*) and exotic common carp (*Cyprinus carpio*) were introduced into these springs at random by various individuals over time, and provided some fishing for youngsters and a few adults. These fish were very difficult to catch, probably because of all the natural food already available.

Every October, after the crops had matured, the irrigation canal gates were closed and all the water from the Comanche Springs was diverted down the old Comanche Creek channel. This was done to allow the irrigation district to clean and repair the concrete canals that delivered water to the various farming areas. This abundance of water flowing down the old historical Comanche Creek bed re-created the historical flow of Comanche Creek and its marsh-like condi-

tions. An astounding number of fall migrating waterfowl was attracted to this wetland each year for the several months it lasted. The flow of water was usually turned back into the irrigation canals by the end of December.

All the springs of the Comanche Springs complex ceased to flow during the great drought of the 1950s. This drought, coupled with the increasing trend of clearing new farmland, and the drilling of irrigation wells that tapped the upstream aquifer of the springs, resulted in the loss of flow of nearly all the springs in

Pecos County by 1957. Even now, if rainfall is abundant over a large area of the recharge region, the springs may flow just enough during the mid-winter months to send a small stream of water down the old creek bed for several months. I believe this last occurred in 1990 - 1991. When the pumps start again for a new growing season, the flow quickly stops. In about five years, the farming situation in central Pecos County shifted from a farming economy based on free flowing spring water, to one based on expensive pumping of water from the ground.

LEON SPRINGS

Leon Springs, located about 8 miles (12.8 km) west of Ft. Stockton, was another large spring complex that was allied with the Comanche Springs complex. The Leon Springs once flowed down its own northward drainage called Leon Creek. It too dwindled away at the same time as the Comanche Springs complex. A general trend by the 1920s was for landowners that had a spring on their property to construct stone and earthen dams across the streambeds below the springs to hold water in a reservoir. Many of these spring-fed reservoirs were stocked with game fish to provide private fishing holes for the landowners and their friends. A large dam across Leon Creek formed a sizable lake, aptly named Leon Lake. The water in Leon Lake was used to irrigate a great many acres of land just north of the lake called the Webb Farms. Today, water pumped from the aquifer below the non-flowing springs irrigates about 100 acres (40.5 ha) of pecan trees on the old Webb Farm lands.

In the 1940s, fishing rights at Leon Lake were restricted to members who paid dues into a fishing club. The fishing was great. The lake was full of largemouth bass and channel catfish. Native fishes were pretty well confined to the upper springs and the canal below the dam, where fisherman seined for baitfish. Mexican tetras were the preferred bait, but roundnose minnows, pupfish and killifish (*Fundulus*) were almost as good. Now I know the correct names of those baitfish, but back then we knew them as shiners, stripers, pigfish and zebras respectively. These same genera were also abundant in the cement-lined irrigation canals below Comanche Springs, and they were the source of baitfish for anyone preparing to go fishing within the region. Comanche and Leon springs had a combined flow of 40,000 acre-feet (49,000,000 cubic meters) in 1946 (Texas Water Development Board, 1984). Today it is zero.

OTHER SPRINGS

Most of the springs in the area were tied into spring complexes interconnected by underground channels of the same aquifer. It has been historically noted that the headwaters of Comanche Springs flowed about 4 miles (6.4 km) down the Comanche Creek bed, where the water disappeared into the ground. Several kilometers below this point a string of at least eight named springs were located, each separated by 6 to 10 miles (9 to 16 km). These springs flowed from beneath rock ledges or gushed forth from what

was described as "deep holes in the ground". Each of these springs formed extensive marshes, but then the water disappeared into the ground downstream just as it did with Comanche Springs water. Muskrats were apparently plentiful in these marshes and in the Pecos River as well (Bailey, 1905).

There are few records of fish at these springs. A local resident who once lived near Casa Blanca Spring is quoted as saying that as a young boy he caught fish

on many a morning from the creek fed by Casa Blanca Spring for his family's breakfast (Adams, 1984). The kinds of fish caught were not mentioned.

The only spring of this chain I had any personal experience with was San Pedro Springs. This spring was deep and some of the channels leading from it were deep and wide. The San Pedro Land Company owned the land and irrigated several hundred acres (81-120 ha) of land by irrigation ditches. San Pedro Springs was a great fishing hole in the 1940s for largemouth bass and several of the smaller centrarchid species. The landscape around the springs greatly resembled the Diamond Y Spring area today. I cannot recall any small, non-game fish present at San Pedro Springs. All these springs have been dry since the late 1950s. It is interesting to speculate what species of fish might have occurred in this string of springs in the 1800s.

Leon Creek had a few small springs along its course, but the Diamond Y Spring system was the largest, and consisted of a large head spring and several smaller springs along its creek bed. The Diamond Y main spring still flows today, although the flow is reduced from what it flowed in the past. We fished at Diamond Y a few times, but it was never as productive as other fishing holes. The spring itself was not good for fishing, but deep holes of water along the creek bed sometimes produced a nice catch. In the

1980s, I was surprised to find one of my old fishing holes still held quite a bit of water. Drilling for oil in the marshy area below the springs could have proved disastrous in the late 1980s, but fortunately, the damage was minimal.

East of town, Tunas Springs was a favorite hang-out. The Tunas Springs (earlier known as Escondido Springs) consisted of a series of three springs along Tunas Creek. The headspring (or West Spring) issued forth from below a limestone ledge and ran eastward for about 8 km until it disappeared into the creek bed, but it kept a number of deep holes filled with water. Catching fish in these deep holes was sporadic, but it was a beautiful place to while away a summer day, and if fishing wasn't good, hunting arrowheads was. A stagecoach stand was built here long ago close to the spring. During the 1930s, the highway department constructed a roadside park along old U.S. Highway 290 (now Interstate Highway 10) on the hill just above the springs, and built a replica of the stage stand at the park, uphill from its original location. The park is located about 32 km east of Fort Stockton on Interstate Highway 10. Irrigation wells drilled just west of the springs tapped that aquifer too, and the springs have been dry since the late 1950s. The only small fish I remember from this area probably were *Diionda*, stripers in our fish vocabulary of the time.

PECOS RIVER AND TRIBUTARIES

Farther east, the Pecos River and two of its tributaries, Live Oak Creek and Independence Creek, were two of our favorite get-away-from-home-and-campout places to go fishing. The mouths of both these creeks could be real hot spots for catching fish, and great places to romp and play in the water. On my first trip to the mouth of Live Oak Creek, a memorable catch was made. A large freshwater eel was caught on a trotline set in the Pecos River one night. An old river fisherman by the name of Doss seemed to know all about eels. He informed us that he would fry the eel for supper that evening, and if any were left over we would have it for breakfast. However, he said, it would have to be refried for breakfast because fried eel turned raw when it got cold. I believe I tasted it that night,

but not for breakfast. A number of large channel catfish were caught every time we fished that area. Live Oak Creek originates in Sutton County, passes just west of old Fort Lancaster, and empties into the Pecos River just east of Sheffield, Texas. The only fish I have a record of from Live Oak Creek is *Fundulus zebrinus*. These are in the Sul Ross State University (SRSU) collection and they were seined from beneath the Interstate Highway 10 bridge that spans the creek just east of the Pecos River crossing.

Independence Creek lies in Terrell County, and in my memory, it is the most beautiful place within 300 km of the Pecos River. Our whole family made an annual pilgrimage in the summer for a number of

years to camp for a week or more on the Lindsey Hicks ranch on the upper end of the creek. My brother and I were about 12 and 13 when we first made the trip from Fort Stockton. The purchase of the Hicks' Ranch by Pinky Roden ended our family camping trips. Later in life I met Joe Chandler who owned the lower end of Independence Creek. My field zoology classes were always welcome at the Chandler Ranch and the Sul Ross Vertebrate Collection contains numerous biological materials from that area. Here the Trans-Pecos copperhead (*Agkistrodon contortrix pictigaster*) was first discovered by Dr. Frank Blair (Gloyd and Conant, 1943). Later, the first record of barking frogs (*Eleutherodactylus augusti*) from Trans-Pecos Texas was reported from the limestone canyons above the confluence of Independence Creek and the Pecos River (Scudday, 1965). Specimens of the river carpsucker (*Carpionodes carpio*) from the creek and the Pecos River, and a single specimen of the blue sucker (*Cycleptus elongatus*) from the Pecos River just above the mouth of Independence Creek are deposited in the Sul Ross Vertebrate Collection. The creek itself supports a rich fish fauna, including the Rio Grande cichlid (*Cichlasoma cyanoguttatum*). If there was only one place you could visit in west Texas for overall beauty and biological diversity, go to Independence Creek.

Below the mouth of Independence Creek, the Pecos River begins to cleanse itself as it flows through high limestone canyons and picks up additional fresh spring water. There was one place downstream from Independence Creek where we had to lower our camping gear and ourselves by rope down a bluff to the riverbank. Trotline fishing for catfish was very good, and fishing with rods and reels during the day provided a nice catch of a somewhat flat, shiny fish the men called a gaspergou. This was my first encounter with what I later learned is more correctly called the freshwater drum (*Aplodinotus grunniens*).

Another good fishing place on the Pecos River was somewhere between Grandfalls and Imperial. I recall it was a large pool with water moving very slowly through it. My most impressive memory is of numerous gars swimming lazily on the surface of the water. My brother and I thought it looked like a great swimming hole, but the gars were too intimidating. Here was my second experience with catching gaspergou and my first introduction to the white bass (*Morone chrysops*).

PHANTOM LAKE

Perhaps our all time favorite fishing hole was Phantom Lake near Balmorhea. Here the water poured forth from a large cave opening in the hillside to the west, forming a small lake of probably less than 3 ha. There was a low bluff on the north side and the water spread out to the south and east of the bluff. Catfish, largemouth bass and crappie (*Pomoxis*) were the main fish caught, but a variety of sunfish could also be caught. Dad's primary interest was the largemouth bass, but my brother and I concentrated on the crappie. We thought they were the best eating fish of all the fish varieties we had sampled.

As youngsters do, we would get tired of fishing and look for something else to do for a while. When exploring above the bluff, we found a large crevice that seemed to drop into a black hole. It was scary looking, and we decided to bring a flashlight with us the next time we came to Phantom Lake, which we

did. When we shined the light into the hole, we saw there was a large flat boulder just below the opening. After dropping some rocks into the hole, we saw numerous bats flying over the big boulder. We were determined that some day when we got older we would come back and explore that hole with some proper equipment. I guess we were about 13 and 14 then. Three years later, about 1946 or 1947, we had our driver's licenses, the war was over, and young men our age had lots of freedom. My brother and I, with three or four of our high school buddies, returned to Phantom Lake with flashlights, ropes, and leather gloves.

We had no problem getting down onto the flat boulder, off the boulder, and onto the floor of what looked like a dry, underground stream. We went what seemed to be downstream a short distance when we came to a narrow crevice that led off to our left. Be-

yond the entrance of the crevice, we could hear the sound of rushing water. The slenderest one of us decided he would try to follow the crevice. He quickly returned and said he went only about 15 meters when he encountered a running stream. He also stated that the crevice was wide enough for any of us to make it through, and it widened even more when it reached the water so at least two of us could stand there at one time. The most fascinating thing about the excursion was finding that shining the beam of a flashlight into the water attracted great numbers of small catfish right up to you. One could actually reach into the water and pick up a fish. The small fish appeared to be blind, having what appeared to be white membranes over the eyes, yet they could detect the light of the flashlight beam in the water and move toward it. None of the fish appeared to be more than about five to six inches in length, and all had darkly pigmented skin. Our last visit to the site was about 1948.

More recently, I have been back to that area twice, once in the mid-1970s, and again in the 1980s. I thought for sure that I could go right to that opening, but I was unable to locate it. The lake itself has been long gone because of the loss of water volume. Man has altered the area around Phantom Lake Spring over time, perhaps in trying to coax more water from the springs and to discourage SCUBA divers from entering the large spring opening. The first measure was to place a heavy grate over the large opening from which Phantom Spring emerged to keep divers from entering the spring. Several divers have died while trying to explore the origin of the spring. Clark Hubbs recently (November, 2001) informed me that the Reeves County Water Improvement District also filled in the openings thorough which we once entered the dry portion of the cavern, and that Phantom Lake Spring had ceased flowing completely in 1999.

RED BLUFF LAKE

Another of my favorite fishing holes was Red Bluff Lake on the Pecos River along the New Mexico state line. It is difficult to look at Red Bluff Lake today and realize what a great fishing lake this once was during the 1940s. The salt content of the water in the reservoir was much lower than it is today. Some of my earliest memories of Red Bluff are of standing below the spillway of Red Bluff dam and catching fish as torrents of foaming water poured over the spillway and down the Pecos River. I have no idea how long it's been since water has gone over the spillway of Red Bluff dam. Fishing for largemouth bass was always good in the lake itself using the baitfish we brought with us from Fort Stockton. Some men pre-

ferred to grapple for big catfish with their hands by diving under water and feeling beneath the large rocks that formed the riprap on the lakeside of the dam. Probably because of the drought and the decrease of flow from the Pecos River, the lake was getting too salty for freshwater fish by the mid 1950s. Sometime after that, the Texas Parks and Wildlife Department began to experiment with stocking saltwater fish into the lake. As I remember, red drum (*Sciaenops ocellatus*) and striped bass (*Morone saxatilis*) were introduced, and perhaps other species. I never fished Red Bluff after about 1949, but I did hear that the redbfish did quite well for a time.

RIO GRANDE

The Rio Grande was fished for catfish with trotlines, rod and reel, and jug floats. Channel catfish and flathead catfish (*Pylodictis olivaris*) were about all I remember catching. The Rio Grande flowed an abundance of clear water, unless it rained upriver. The fish we caught in the 1940s and 1950s were very good to eat. Channel catfish between 1 and 2 pounds (0.5 and 0.9 kg) were primarily caught in rapids on rod and

reel, while it was not unusual to catch flathead catfish up to 9 kg on the trotlines and jugs. One of the best days catch I made with a rod and reel was about 1943 just upstream from the little village of Ruidosa, Presidio County, Texas. Good rapids were there then from which the village got its name. Today, the Rio Grande no longer flows consistently through that stretch of the river. The river does begin to flow again below

Presidio where water from the Rio Conchos coming in from Mexico reconstitutes the Rio Grande. From this point downriver, the water is so low and polluted with chemicals and sewer effluent, the Big Bend National Park has posted signs warning visitors not to eat any fish they catch from the Rio Grande.

Beavers (*Castor canadensis*) still occur along some stretches of the Rio Grande bordering Brewster

and Presidio counties. Muskrats once occurred along parts of the Pecos River and in the marshes of the Comanche, Leon, and San Soloman spring complexes (Bailey, 1905). The Pecos River muskrat (*Ondatra zibethicus ripensis*) was described by Vernon Bailey in 1902, and may still exist today in irrigation drainage ditches in El Paso County and possibly southeastern New Mexico (K. Holmes, 1970; J. Holmes, 1970).

CONCLUSION

Now, in the year 2001, another devastating drought has plagued this region since 1992, a drought I consider worse than the one in the 1950s. Ongoing development of massive pump-irrigation projects and the acquisition of water rights by large cities continues at a rapid pace in this desert region. Water levels in underground aquifers are dropping rapidly. The Rio Grande no longer flows into the Gulf of Mexico. The few springs that survived the 1950s drought are now

disappearing. The U.S is threatening to sue Mexico for water owed the U.S., and Texas and New Mexico have continued to do court battle over the water of the Pecos River. Will we see more drastic water wars in the future? Given current trends, and based on personal observations of declines during the past six decades, the eventual loss of the remaining Chihuahuan Desert fishes seems inevitable.

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AQUATIC CONSERVATION AND THE NATURE CONSERVANCY IN WEST TEXAS

JOHN KARGES

ABSTRACT

Aquatic biodiversity in the Chihuahuan Desert portion of West Texas is high and of increasing conservation concern because of the high incidence of endemism, limited range distributions for both species and natural communities and the limited areas where surface waters still occur. Surface waters include remaining intact or restorable reaches of principal river systems, perennial or permanent-pool streams, and isolated springs with their attendant outflows and marshes. Some aquatic sites are well known and conserved by either government agencies or private conservation organizations, while other areas are either not protected or perhaps not even identified. Over the last decade, The Nature Conservancy has invested considerable money, time and resources in conserving specific, critical areas harboring rich aquatic biodiversity in the northern Chihuahuan Desert by purchasing preserves and partnering on adjacent lands

with conservation easements to provide permanent protection of rare aquatic areas. The Conservancy's protection and conservation efforts include identifying and mapping the distribution of rare aquatic species, assemblages and communities within a landscape context (ecoregional planning), long-term landsite protection and research, monitoring and stewardship management on the site and its biological elements. In addition, research and planning for groundwater issues of depletion and delineation of watersheds and recharge zones is crucial to sustainable, long-term conservation of these imperiled systems (site conservation planning). The Nature Conservancy presently is involved at five sites in West Texas with aquatic conservation elements, and through ecoregional planning will identify additional "action sites" to protect the aquatic biodiversity of the Chihuahuan Desert.

INTRODUCTION

A recurrent and contemporary conservation issue throughout deserts involves aquatic biota and the impacts of declining surface water availability on populations, species, assemblages of organisms or ecological function and integrity of entire systems (Rinne and Minckley, 1991). Throughout continental aridlands, surface waters frequently harbor endemic, rare or limitedly distributed species and in many cases, unique aquatic natural communities. The water may be flowing or stationary and either ephemeral or perennial, depending upon climatic cycles, seasonal durations, topography and underlying geology. Surface water bodies of springs and seeps, creeks, rivers, marshlands and playa basins are widely scattered across the northern Chihuahuan Desert landscape in Trans-Pecos Texas. With gradual but significantly increased aridification of the Chihuahuan Desert region since the

end of the Pleistocene, surface waters have been greatly reduced. Consequently in many cases of isolated springs and perennial, relictual stream reaches, organisms isolated from wider, formerly contiguous distributions have differentiated into distinct genetic entities, in some instances at the full species level, but often at least to recognizable populations or subspecies level.

The rapid diminution and sometimes total loss of surface water in the past century due to a combination of climatic and anthropogenic factors is an additional and much more contemporary threat to the desert's aquatic biodiversity. Aside from Chihuahuan Desert annual or seasonal droughts (typically from mid-winter to mid-summer), relatively long-term droughts of variable duration (up to nearly a decade in some cases),

have occurred periodically since the inception of record keeping in the late 1800s. This coupled with increased human demands and water extraction have contributed to the measurable depletion of most surface spring systems in the Chihuahuan Desert (Hubbs, 1995), and has even included total elimination of some historically robust springs and outflows (e.g., Comanche Springs at Fort Stockton, Pecos County, see also Scudday, this volume). The threat to biodiversity by water reduction and over-harvest has already been manifested in the extirpation of the Comanche Springs pupfish (*Cyprinodon elegans*) at the type locality (U.S. Fish and Wildlife Service, 1981). In fact, within the Chihuahuan Desert portion of Texas, 56.4 percent (22) of the 39 total native fish species are of conservation concern as endangered, threatened or declining by the Texas Organization for Endangered Species (Edwards et al., 1989). An additional human-induced threat to regional aquatic biodiversity is the introduction of competitive exotics, which can seriously impact or degrade ecological integrity and even species' genetic purity through hybridization (Hubbs, 1990).

Among the types of systems and surface features containing identified "hotspots" of aquatic biodiversity within the northern Chihuahuan Desert are relatively intact or potentially restorable reaches of the major rivers, some remaining perennial or permanent pool streams fed by either springflows or runoff drainage from mountainous basins; and desert springs and their associated streams and marshes (ciénegas). The major river system of the region is the mainstem of

the Rio Grande (Río Bravo del Norte) and its significant tributaries, the Río Conchos in Chihuahua, the Pecos River of New Mexico and Texas, and the Devils River entirely within Texas. Perennial streams are scarce and widely distributed. Some of the tributaries of the Devils and Pecos rivers in the eastern Trans-Pecos are spring-fed from aquifers in the limestone matrix of the Stockton Plateau. This is an ecotonal region of plants, animals and biotic communities between the eastern Chihuahuan Desert and the western Edwards Plateau. Farther west in the Trans-Pecos, streams in Big Bend National Park and Big Bend Ranch State Park contain permanent reaches of surface water, fed by subirrigated flow and augmented by periodic precipitation related episodes of flash-flooding. These surface pools and runs can be extremely variable in size, depth, distribution and extent, but as long as some permanent water remains throughout the year, refugia populations of aquatic species persist through seasonal droughts and even flourish during pluvial cycles and events. The last category of surface water types is the remaining desert springs and their attendant outflow runs and marshland systems (Hubbs, 1995) that typically harbor endemics or extremely limited range taxa. These spring systems often are individually distinctive in water volume and chemical parameters. Considered cumulatively across the desert, they include a wide range of dissolved mineral contents, thermal conditions and regimes and even subterranean aquifer linkages between other nearby springs.

HISTORY AND FOCUS OF THE NATURE CONSERVANCY IN AQUATIC CONSERVATION IN WEST TEXAS

Since 1990, The Nature Conservancy has operated a West Texas Program and office within the region, primarily to facilitate stewardship, administration and other organizational functions. The Nature Conservancy's conservation actions are predicated on its mission "to preserve the plants, animals and natural communities that represent the diversity of life on Earth by protecting the lands and waters they need to survive" and specifically within the region, the biodiversity of the Trans-Pecos and northern Chihuahuan Desert. Because much of the endemism and biological richness of the desert occurs at aquatic sites, the Conservancy focuses its conservation attention and resources

on remaining springs and creeks with rare or endemic species and intact ecological function. Typically, sites were identified through the Natural Heritage Program database system of cataloguing and mapping biodiversity, specifically rare species' occurrences, across each state. Several of the conservation sites for The Nature Conservancy within the Trans Pecos region are established primarily for preservation of aquatic resources and systems, including endemics. These sites may be either the Conservancy's preserves owned in fee-title or private partnership lands upon which the Conservancy holds conservation easements with protective but consensual legally binding deed

restrictions. At some sites (see specific details below), notably Independence Creek and the Devils River, the Conservancy uses a combination of fee-ownership and easements on lands owned by others to affect conservation across a broader spatial scale than

could be done by using only a single conservation tool. At others, The Nature Conservancy owns the preserve entirely, for example, Diamond Y Spring Preserve and Sandia Springs Preserve.

ECOREGIONAL PLANNING FOR THE CHIHUAHUAN DESERT

In the past few years, The Nature Conservancy has refined its conservation vision to focus on ecologically functional conservation areas within ecoregions (The Nature Conservancy, 2001). A functional conservation area includes either focal species, natural communities or entire ecological systems, and the supporting ecological processes necessary to sustain them over the long term (Poiani and Richter, 2000). This planning effort is called ecoregional planning and a designed conservation portfolio represents the full distribution and diversity of conservation elements (including native species, natural communities and ecological systems) within the ecoregion. Ecoregional planning is designed to maximize conservation of biodiversity within and across ecoregions, while optimizing critical resource allocation toward site-specific conservation actions by the Conservancy and its conservation partners.

The basic steps in ecoregional planning are: 1) identifying the species, communities and ecological systems, 2) setting specific goals for the number and distribution of these conservation elements to be captured by the portfolio, 3) assembling information and relevant data on the location and quality (contempo-

rary status) of conservation elements, 4) designing a network of conservation areas that most effectively meets the goals, and 5) selecting the highest priority conservation areas in the portfolio for the Conservancy's action and involvement.

Although The Nature Conservancy has just initiated the Chihuahuan Desert ecoregional planning process, most of the aquatic elements of conservation concern (species, assemblages and communities) have been identified and many occur at what will be unequivocal portfolio sites for conservation action. The sites where the Conservancy is currently involved within the ecoregion reaffirm that early assessments about where biodiversity occurs on the land (and waters!) and past decisions about whether the Conservancy should work there, were in alignment with what is now termed ecoregional planning and "Conservation by Design". Future Conservancy "action sites" will undoubtedly include additional intact perennial or permanent pool stream segments and more mountain ranges with isolated springs with aquatic endemics or high composite conservation values based on species and communities.

DIAMOND Y SPRING PRESERVE

This 607-ha (1,502-acre) preserve, located about 20 km NNW of Fort Stockton, Pecos County, contains the sole naturally occurring population of the endangered Leon Springs pupfish (*Cyprinodon bovinus*). Originally described in the 1850s from the now obliterated Leon Springs area due west of Fort Stockton, the pupfish was not substantiated for nearly a century and was presumed extinct. Then in 1965, W. L. Minckley and W. E. Barber collected Leon Springs pupfish from Diamond Y Draw (see Echelle and Miller, 1974 for the historical account). With the rediscovery

of the pupfish, academic and agency attention focused on the Diamond Y Spring area, and the importance of the site soon was found to include other aquatic taxa as well, including the federally endangered Pecos gambusia (*Gambusia nobilis*) and an obscure species of salt-tolerant sunflower (Pecos or puzzle sunflower, *Helianthus paradoxus*) growing in permanently hydric soils along the stream course. In addition, three species of aquatic and littoral zone snails also occur at the preserve. Two are endemic, the Diamond Y spring snail (*Tryonia circumstriata*, formerly *T. adamantina*,

Hershler et al., 1999) and the Gonzalez spring snail (*Tryonia stocktonensis*). The third species, Pecos assimineia (*Assimineia pecos*) has a wider distribution but is still essentially endemic to the middle Pecos River basin and was recently proposed for listing as federally endangered. Other groups that may be represented as endemics or at least genetically distinct forms in the Diamond Y Spring system include the amphipod genus *Gammarus* and an undescribed crayfish, both of which are under investigation currently.

Diamond Y Spring is a solution cavity in the middle of an alluvial basin with a stream outflow into an alkaline marshland. The stream flows across the preserve to the confluence with Leon Creek which ends there as a named topographic feature. Diamond Y Draw continues northeastward through the preserve and across Pecos County to the Pecos River. There are two primary stretches of surface water with the rare, aquatic species, one from the spring to and beyond the Leon Creek confluence, and a second reach about 2 km downstream to the north. Both reaches are augmented by small peripheral springs and seeps along their borders, and some of these harbor the endemic invertebrate fauna.

The Nature Conservancy acquired Diamond Y Spring Preserve in 1990, and began to initiate on-site stewardship almost immediately. The site is in an actively producing oil and gas field. Energy production companies have been partners by providing supporting funds for the purchase of the preserve and helped with certain safeguards for the protection of the surface waters from any contaminants that had threatened or plagued the site in the past. The measures included decommissioning buried corrosible metal pipelines in areas adjacent to vulnerable aquatic resources and their replacement with synthetic surface lines that could be easily monitored and repaired if necessary, as well as emergency shut-off valves installed at both sides of any creek crossings. At production sites, oil well pads were bermed to sufficiently contain any potential contaminant spill volume prior to detection. A matching grant in the mid-1990s from an energy producer and the National Fish and Wildlife Foundation provided the mechanism to remove some abandoned pad sites and their raised access roads from Diamond Y Draw which actually had impeded surface water flow through the marshland.

An urgent conservation issue arose when genetic contamination of the Leon Springs pupfish was detected with measurable introgression and both ecological and genetic competition from the introduced congeneric sheepshead minnow (*C. variegatus*) (Echelle and Echelle, 1997; Echelle et al., this volume). This genetic contamination had been addressed at Diamond Y Spring in the mid-1970s (Hubbs et al., 1978), but apparently persisted even after an intensive attempt to eradicate the exotic sheepshead minnows and hybrids. Another effort to eradicate the sheepshead minnow genome was initiated in the late 1990s and is presently ongoing. Investigators representing the Rio Grande Fishes Recovery Team outlined and implemented a restoration plan. The plan's sequential actions addressed the hybridization and competition issues through elimination of introgressed genomes and replacement with genetically pure Leon Springs pupfish stock from the reserve population held at Dexter National Fish Hatchery in Dexter, New Mexico. Preliminary results on the effectiveness of the genetic restoration are promising (A. A. Echelle, Oklahoma State University, pers. comm.). For additional discussion on the conservation and ecology of the Pecos gambusia and the elimination of the competitive exotic largespring gambusia (*Gambusia geiseri*), in the Diamond Y Spring system see Hubbs (this volume and Echelle et al. this volume).

Another conservation concern of paramount urgency is the pervasive threat to groundwater availability and spring discharge that sustains the suite of endemic and rare obligate aquatic species and communities at Diamond Y. Although the Conservancy has been effective at securing the immediate land around the spring and the watercourse with surface water, the issue of recharge and discharge volume is much larger than the presently protected land-base. Topographically, Leon Creek enters the property with minimal (and sporadic) surface runoff and some subirrigated flow through the valley alluvium. The creek headwaters and surface drainage basin begin at the now dry Leon Springs west of Fort Stockton and courses northeastward for approximately 16 km before entering the preserve. Surface flow of Leon Creek is ephemeral and seasonally episodic in the last km before the confluence with Diamond Y Draw, which does have permanent discharge from Diamond Y Spring and peripheral seeps (Veni, 1991). Surface water presently

only extends for perhaps another one km before disappearing in the alluvium. A. A. Echelle and S. E. Kennedy (pers. comm.) recall much more extensive surface flow during the 1970s, including relatively broad standing water “flats” downstream, depicted in Kennedy’s (1977) map of the upstream segment. Emergent flows again appear on the surface approximately 2-3 km downstream with peripheral spring discharge supplementing the subirrigated alluvial flows in the valley floor at the northeastern end of the preserve.

The Conservancy and its conservation partners from agencies and academia face a tremendous challenge with the problematic perpetuation of the spring discharge and therefore the sustenance of the aquatic system and species. Groundwater recharge and aquifer interactions still are imperfectly defined for the entire area (Veni, 1991; Boghici, 1997; Boghici and Van Broekhoven, 2001), and long-term measures to ensure that discharge is sufficient to maintain the system’s functional integrity and conservation values are not in place. There are crucial adjacent tracts the Conservancy should protect at the immediate preserve boundary but this does not adequately protect a system fed

by an ostensibly large but beleaguered off-site aquifer and potentially huge recharge zone that likely covers hundreds of square kilometers. These broader groundwater issues will be addressed through the Conservancy’s Site Conservation Planning process, a formal assessment of the conservation elements and threats with a strategic plan for permanent and effective conservation as well as through species specific Recovery Plans. However, each of these planning efforts will have to take on the complicated and controversial issue of groundwater that will have far-reaching political and economic implications.

The Conservancy’s on-site stewardship has included aggressive salt cedar (*Tamarix ramosissima*) and mesquite (*Prosopis glandulosa*) control along the watercourse to reduce water consumption by these invasive species, both of which are relatively voracious water users. With the addition of prescribed fire, mechanical and judicious chemical treatment of these trees, this effort is to reduce as much superfluous water consumption as possible and retain it in the surface water segments of Diamond Y Draw.

SANDIA SPRING PRESERVE

At the northeastern outwash plain of the Davis Mountains is a complex, interrelated system of springs that include Phantom Lake, San Solomon, Giffin and Sandia springs as well as some relatively smaller seeps. Phantom Lake Spring is of grave conservation concern currently with virtually no outflow and consequently critical endangerment to the suite of aquatic rarities found in the immediate cave entrance, including Comanche Springs pupfish, Pecos gambusia and the invertebrate assemblage of snails and amphipods. San Solomon Spring in Balmorhea State Park has a strong discharge still, although heavily manipulated into a recreational impoundment and channelized outflows serving as pupfish refugia including a flow-through demonstration ciénega for public interpretation. Another portion of the spring complex just east of Balmorhea is East and West Sandia springs on 97 ha (240 acres) owned by the Conservancy. West Sandia Spring is a tiny spring, often with no apparent flow. It emanates from a cavity in the alluvial valley floor and the outflow channel is only several hundred meters in

length with no known connection to either a larger drainage or even the irrigation canal system that surrounds it. Currently, no species of conservation concern are known from this system. However the larger East Sandia Springs, located several hundred meters to the east, does harbor species of concern including snails, amphipods, puzzle sunflower and Pecos gambusia with historical occurrences of Comanche Springs pupfish. It also contains aggressive native predatory species like green sunfish (*Lepomis cyanellus*), as does the San Solomon system, but they are relatively inconsequential to the rare fish species (Hubbs, 1993).

At Sandia Springs, the critical issues are the same as those facing all of the other springs in the Balmorhea complex as well as at Diamond Y Spring. Exotic saltcedar invasion draining shallow groundwater reserves and the possibility of sheepshead minnow genetic contamination of pupfish are two localized threats to the system’s integrity. The mature saltcedars have been removed mechanically but rapid and recurrent

recruitment will require punctual, intensive and continuous control. The overarching conservation dilemma for the entire Balmorhea Spring complex is diminished spring flows, delineation of the recharge zone and replenishment rates of aquifer(s) that contribute

to surface discharge. Until these questions are answered and comprehensive threat abatement strategies are designed and implemented, most localized and proximate land conservation efforts are myopic stop-gap solutions at best.

INDEPENDENCE CREEK MEGASITE

The Pecos River is divided into two distinct segments in Texas from its entry at Red Bluff Reservoir on the New Mexico border to the confluence with the Rio Grande. Across the Pecos Plain (Permian Basin), the Pecos River is a slow, meandering river impacted by dewatering from diversions, groundwater withdrawal, saltcedar infestation and erosion (Hoagstrom, 2000). There is virtually no supplementary inflow from tributaries or springs. After entering the limestone canyonlands it has carved downstream of Iraan, it is augmented by very few perennial streams and some relatively small springs. Of these contributory sources, one of the most important in terms of both water quality and volume is Independence Creek, which merges with the Pecos River in Terrell County, approximately 45 km south of Sheffield.

Independence Creek has long been recognized as a biologically diverse site because of strong perennial instream flow and its riparian corridor. Ichthyologists have heralded Independence Creek as a clear, spring-fed refugium for the native Pecos River fish fauna, particularly as a repopulating source of the Pecos River following episodic and lethal mass fish mortalities resulting from red tides related to dinoflagellate (*Prymnesium parvum*) blooms. These recurrent outbreaks periodically decimate fish populations along stretches of the Pecos River. Independence Creek and only a few other smaller freshwater tributaries between Iraan and the Rio Grande confluence at Lake Amistad are reestablishment sources as native fishes return to the river after the effects of the red tide blooms have abated (Rhodes and Hubbs, 1992). The assemblage of native Pecos River fishes of conservation importance includes the Rio Grande darter (*Etheostoma grahmi*), proserpine shiner (*Cyprinella proserpina*), headwater catfish (*Ictalurus lupus*). The shiner and darter recently have been confirmed in Independence Creek (Valdes, 1994; Karges, field notes, 2000-2001).

Independence Creek is only 16 km in length from its outflow source spring to the confluence with the Pecos River, although the catchment basin extends northwestward nearly to Fort Stockton in Pecos County. The site is called a megasite because of the Conservancy's recognition that it is not sufficient to conserve just a small area immediately around the creek but also an imperative opportunity to work at the landscape scale on a large and significant watershed which affects the creek's functional integrity. The Nature Conservancy's conservation actions at Independence Creek began with the 1990 acquisition of a 284-ha (702-acre) conservation easement with private landowners along the last 3.2 km of the creek including the Pecos River confluence. The easement is structured principally to eliminate degradation of the creek and adjacent riparian corridor, and to retain as much instream flow as possible. In 2000, the Conservancy acquired an 3517-ha (8,690-acre) ranch immediately upstream from the easement property, with the primary purpose of protecting an additional 4.8 km of surface water. The ranch also contains Caroline Springs, a significant, large volume instream source supplementing the creek. However, the springs have been impounded into two relatively large recreational lakes. These lakes, when combined with an additional source of evaporative loss through a large-scale aerial sprinkler array for lawns and channelized irrigation network for improved pastures, precludes much of the spring waters from rejoining the Independence Creek mainstem. Also, introduced large, predatory fishes including largemouth bass (*Micropterus salmoides*) and channel catfish (*Ictalurus punctatus*) in the reservoirs, could hinder the persistence of the native fishes.

The Conservancy is designing a water budget monitoring strategy for assessing water losses between the spring and the creek, with the goal of redirecting

water for conservation of the target species and restoring the strength and integrity of the creek. An additional acquisition of the next upstream ranch adds another 4.4 km of Independence Creek under the Conservancy's management and stewardship, leaving only the remaining 3.2 km from the headwaters without conservation protection. Another stewardship ac-

tion to restore instream water volume along the managed reach of Independence Creek is the initiation of an aggressive saltcedar eradication campaign and reduction of the native, but pervasive, false willow (*Baccharis neglecta*), both of which rob water from the creek.

DOLAN FALLS/DEVILS RIVER MEGASITE

The Devils River is the next major Rio Grande tributary east of the Pecos, in the overlap zone between the Chihuahuan Desert to the west, the Edwards Plateau to the east and the Tamaulipan Thornscrub ecoregion to the south. This system is defined by a catchment basin that is relatively small on a continental scale. It is of considerable significance for both terrestrial and aquatic conservation elements that represent the biodiversity where these three ecoregions converge. The surface water of the Devils River extends for approximately 90 km from its headwaters to Lake Amistad on the Rio Grande, entirely within Val Verde County. The river is generally characterized by long flat-water pools of varying depths with slow-moving currents punctuated by broad, shallow riffles, stair-step cascades, and channelized flow constrained within the fluted bottom topography of the limestone bedrock. Just above Dolan Falls, Dolan Creek joins the Devils River with a substantial spring-fed contribution as the largest tributary along the river, although there are numerous spring-fed smaller rivulets and basal springs along much of the river between the headwaters and Lake Amistad. Neither dams nor pollution have yet impacted the Devils River, and it remains the most intact free-flowing river in West Texas (Harrell, 1978).

Texas Parks and Wildlife Department initiated the first conservation land acquisition along the river with a purchase of about 7,689 ha (19,000 acres) in the 1980s creating the Devils River State Natural Area. The purchase included 2 km of riverfront and the headwaters of Dolan Creek. Subsequently, The Nature Conservancy purchased the Dolan Falls Ranch, about 7,284 ha (18,000 acres) in 1991, protecting Dolan Creek from the headwaters to the confluence. This also encompassed land on either side of Dolan Falls, including nearly 6.4 km of riverfront on the Devils River.

The integrity of the hydrology and its diverse biota in the contributing springs, creek and river has been the primary focus of both entities' land protection efforts. In 2000, the Conservancy also acquired the 8,903-ha (22,000-acre) Devils River Ranch downstream from Dolan Falls adjacent to Lake Amistad National Recreation Area, protecting an additional 19-km segment of the river that harbors the native fish fauna and other aquatic elements. All of the land conservation activity in this area includes terrestrial species and communities as well as abating local threats of development and subdivision within the watershed. The Nature Conservancy has developed a site conservation plan for the entire megasite, which includes much of the watershed but focuses specifically on the surface waters and immediate drainages of the surface water portion of the Devils River.

The formation of Lake Amistad in the late 1960s - early 1970s has impacted the fish fauna of the Trans-Pecos portion of the Rio Grande basin including its Pecos River and Devils River tributaries. Inundation of the confluences as the lake filled fragmented the contiguous distribution of the native guild of river-adapted fishes. Subsequent introduction of exotic predatory species such as striped bass (*Morone saxatilis*) and smallmouth bass (*Micropterus dolomieu*) has likely contributed to the decrease of some native species to some extent. Three native fish species are considered extirpated within the system and it is unlikely that residual populations or remaining suitable habitats exist in any peripheral areas. Most of the rare species of this riverine fauna require the flowing waters found in the Pecos and Devils rivers (Valdes Cantu and Winemiller, 1997). Three other rare fishes are associated with flowing waters of the Devils River, including proserpine shiner, Devils River minnow (*Dionda diaboli*) and Rio Grande darter, and popula-

tions have been confirmed in the Devils River recently (Garrett et al., 1992). Headwater catfish have also been confirmed in the last two decades from both the lower Pecos River (Kelsch and Hendricks, 1990) and Devils River system (Dolan Springs/Creek in 1980, Garrett et al., 1992). This species is morphologically very similar to the widespread and frequently introduced channel catfish, but is known principally from clear, spring-fed systems.

The Devils River below Dolan Falls is primarily lentic with only short reaches of swift water, riffles and limited suitable habitat for fishes requiring those conditions (Harrell, 1978). The local pupfish has been identified as the Conchos pupfish (*Cyprinodon eximius*) but may represent a disjunct, undescribed form that is endemic to the Devils River above the lake (Hubbs et al., 1991). The pupfish has been reestablished in Dolan

Creek through introduction from populations in the Devils River (Hubbs and Garrett, 1990) and may be found in lentic pools along the Devils River downstream to at least Pafford's Crossing (Davis, 1980) and Big Satan Canyon (Karges, unpublished field notes). An additional aquatic species that may prove to be an undescribed endemic is a neotenic salamander in the genus *Eurycea*. A few specimens are known from springs along Dolan Creek.

Most of the on-site stewardship and monitoring actions related to aquatic conservation at Dolan Falls Preserve have been the continuation of periodic surveys of the fish fauna (Valdes Cantu and Winemiller, 1997; G. P. Garrett, Texas Parks and Wildlife Department, pers. comm.), aquatic invertebrate inventories, and in redirecting visitor use and foot traffic at the fragile micro-habitats at the spring outflows.

MADERA CANYON PRESERVE

Within the Davis Mountains, few perennial streams remain and of these, Little Aguja Canyon (Jeff Davis County) on the northern slopes is among the most important because of the presence of two aquatic species. The federally endangered Little Aguja pondweed (*Potamogeton clystocarpus*) is a cryptic species known only from this canyon, in plunge pools and subirrigated permanent pools along the middle and upper reaches of the drainage. Also in these permanent pools, the Rio Grande chub (*Gila pandora*) survives as the only relict population remaining in Texas (Miller and Hubbs, 1962). Although the species has not been collected in 16 years, 4 individuals were seen in a pool on the Conservancy's Madera Canyon Pre-

serve in 2000 (Karges, unpublished field notes) and likely remains in other similar pools throughout the upper and middle sections of the canyon. The Nature Conservancy includes this area in the overall site conservation plan for the Davis Mountains which includes other rare aquatic species including Davis Mountains spring snail (*Fontilicella davisii*) (Taylor, 1987) and the possibility of reintroducing Rio Grande cutthroat trout (*Oncorhynchus clarki*) to Davis Mountains streams (Garrett and Matlock, 1991) if some highland streamcourses can be restored to perennially flowing montane systems with landscape scale watershed management to restore recharge and instream flows.

DEDICATION

This paper and my conference presentation are dedicated to the memory of Dr. W. L. Minckley for his contributions to desert fish research and conservation. Also, during a Chihuahuan Desert conference

in Monterrey, Nuevo Leon (the only time we ever met), he recounted the delightful and intriguing story of his role in the rediscovery of the "extinct" Leon Springs pupfish.

ACKNOWLEDGEMENTS

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fieldwork and sharing their time, knowledge and exuberance for desert fish diversity and its conservation.

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INNOVATIVE APPROACHES TO RECOVER ENDANGERED SPECIES

GARY P. GARRETT

ABSTRACT

Texas Parks and Wildlife Department has embarked on a progressive approach to resolving endangered species issues through involvement with local governments and especially private landowners. Work in the Balmorhea area of West Texas involved local citizens, the City of Balmorhea, three universities, four NGOs, five state and three federal agencies. Together we created a natural ciénega, made progress towards eliminating a source of genetic contamination, enhanced nature tourism and began developing an improved sport fishery. We are now working on Conservation Agreements in West Texas that will enable the resolution of concerns over two candidate fish species and, if suc-

cessful, will preclude the need to list these species as endangered. With the Devils River minnow, we are working closely with landowners and the City of Del Rio to determine and resolve life history requirements and restore populations to natural levels. In so doing, the cooperating entities will also be protecting the quality of the Devils River and associated streams. A major component of the Pecos Pupfish Conservation Agreement is to create alternate habitats on private land in Texas. By involving individuals and local governments, we are more likely to achieve long-term benefits for natural resources as well as public health and quality of life.

INTRODUCTION

Approximately 25% of the 169 native freshwater Texas fishes are of conservation concern (Hubbs et al., 1991). That is, they are either in danger of extinction or approaching that status. In the Chihuahuan Desert region of Texas, 50% of the native fishes are of conservation concern or already lost to extirpation or extinction (Hubbs et al., 1991). When species are being lost, their predicament is usually indicative of a larger biological problem. Reversing the trend and avoiding extinctions is usually complex, politically volatile and expensive.

Texas Parks and Wildlife Department (TPW) is working with federal, state and local agencies and es-

pecially private landowners to conserve natural resources and resolve endangered species issues. With 97% of the land in Texas privately owned, involvement with the private sector is often the only way to achieve long-term conservation goals.

Herein, I present three case histories of innovative approaches to resolve specific issues. Although each situation has unique elements, these case histories serve to provide examples that have worked and to generate new ideas on how to approach such issues.

COMANCHE SPRINGS PUFFISH AND SAN SOLOMON CIÉNEGA

In 1996, a cooperative effort among private, state and federal entities allowed the creation of a desert-wetland (ciénega) habitat for two federally endangered fish species, Comanche Springs pupfish (*Cyprinodon elegans*) and Pecos gambusia (*Gambusia nobilis*). The primary benefit to the fishes is a "natural" habitat criti-

cal to their survival, but which had been eliminated through human modifications for recreation and agriculture. Benefits to area residents included: 1) relaxation of some pesticide regulations for farmers, 2) protection of the water supply and 3) increased tourism.

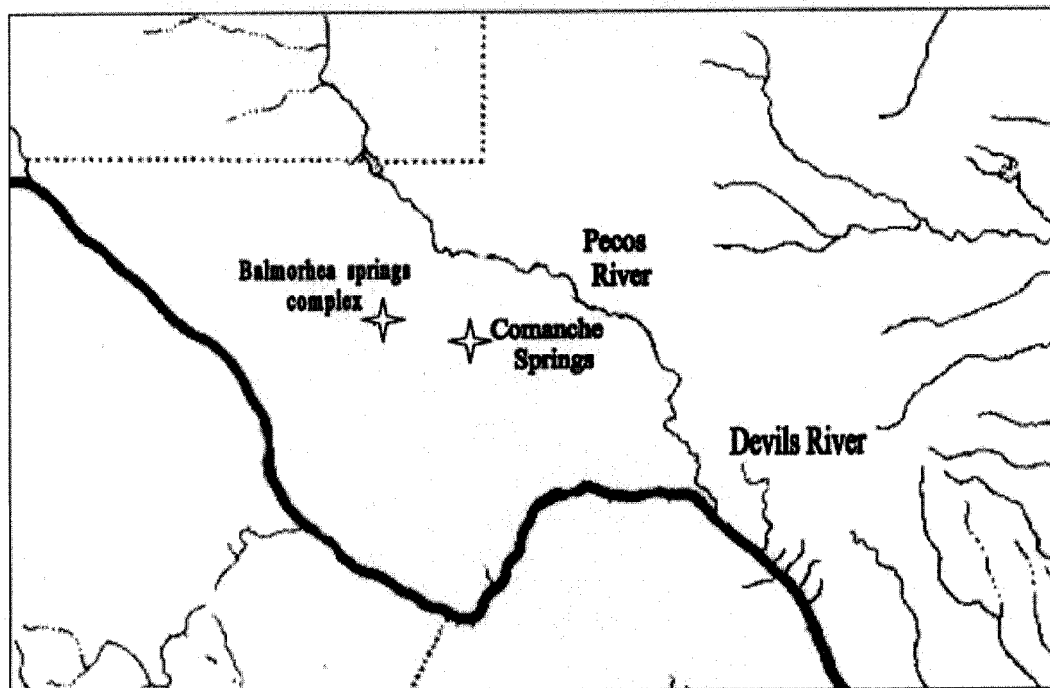


Figure 1. Project locations in West Texas.

Prior to human alterations, Comanche Springs pupfish and Pecos gambusia inhabited two large *ciénega* systems separated by approximately 100 km (Figure 1), one fed by the Balmorhea springs complex (Phantom Lake, San Solomon, Giffin and East Sandia springs), and one by Comanche Springs. San Solomon Springs is the largest spring in the Balmorhea springs complex, producing about 750 liters per second (lps); it is presently the largest spring in the Trans-Pecos and the seventh largest in Texas. Comanche Springs, in Fort Stockton, once flowed at approximately 1,200 lps, but ceased its perennial flows in 1961 due to groundwater pumping (Brune, 1981; Scudday, this volume).

More than 100 farmers depended on surface, irrigation waters flowing out of Comanche Springs and the *ciénega*. Groundwater pumping by 18 land-owners in an area west of Fort Stockton severely diminished the flows from the spring. The local water district sued these pumpers in 1952 in an attempt to

establish their water rights. The pumpers prevailed in the lawsuit by basing their defense on a 1904 case from which had emerged the concept of "right of capture." This concept established that a well owner could pump as much water as desired, regardless of the impact on the aquifer. This was also the case in which the Texas Supreme Court had determined that the intricacies of aquifers were so "secret, occult, and concealed" that it would be impossible to administer a set of rules. Ultimately the flows of Comanche Springs ceased, the *ciénega* dried up, the native flora and fauna disappeared, the surface irrigators lost their farms and their land reverted to desert.

Similarly, farmers also diverted water for agriculture from the Balmorhea springs complex and have been doing so since the mid-1870s (Brune, 1981). In 1915, the Reeves County Water Improvement District No.1 (RCWID) was established and, with water from San Solomon and other associated springs, administered irrigation water for 4,900 ha of farmland.

Ciénegas presumed to have supported large numbers of *C. elegans* and *G. nobilis* were drained and spring flows were diverted into an irrigation network of concrete-lined canals with swiftly flowing water and dredged, earthen laterals. This habitat is highly unnatural, ephemeral and wholly dependent upon local irrigation practices and other water-use patterns. In the 1930s, the Civilian Conservation Corps modified San Solomon Springs into a large swimming pool at Balmorhea State Park. The work of this New Deal program enhanced the park's visitor services, but further disrupted the natural ciénega.

Ciénegas, and their associated springs, provide habitat for a wide variety of plants and animals, some of which are endemic to these systems (Hendrickson and Minckley, 1984). Not only can ciénegas harbor unique species, but also an entire community of interacting organisms depends on these fragile habitats for survival. This is especially true for the increasingly rare desert fishes. Few ciénegas have survived intact to this day. When the original San Solomon ciénega was modified, and for the most part destroyed, the only "aquatic habitat" remaining was in the concrete irrigation canals. Although better than no habitat at all, the irrigation canals, at best, provided a tenuous existence for some life forms. Some indigenous species, such as the Pecos River muskrat (*Ondatra zibethicus ripensis*), did not adapt and were extirpated. The Comanche Springs pupfish and Pecos gambusia managed to survive in the irrigation canals, but their numbers were greatly reduced. Because of the loss of most of their natural habitat, both fishes are rare and on the federal and state Endangered Species lists.

Previous efforts to improve habitat have occurred in the Balmorhea area. A small refugium canal (120 m) was constructed in 1974 at Balmorhea State Park (Echelle and Hubbs, 1978). During a two-year sampling study (Garrett and Price, 1993), Comanche Springs pupfish population size in the park refugium canal was estimated to be as low as 968 (May 1990) and as high as 6,480 (September 1990).

In 1993, a modified canal was constructed at Phantom Lake Springs by the Bureau of Reclamation (Young et al., 1994). Instead of the original concrete walls, the 110-m canal has sloped, earthen, sinuous sides and was designed to resemble a portion of a

ciénega. The Phantom Lake Springs Refugium Canal resulted in an increase in local abundance of Comanche Springs pupfish, resulting in an average of 14.7 pupfish/m² (Winemiller and Anderson, 1997). Unfortunately, the springs have now failed (Hubbs, 2001) and a pump is needed just to maintain water in a small pool at the spring source (N. Allan, U.S. Fish and Wildlife Service, pers. comm.).

People also suffer when their water sources vanish. Farmers who depended on surface irrigation water from Comanche Springs lost their livelihood when the springs went dry. Farmers in the Balmorhea area also rely on surface irrigation from springs, and if the aquifer were diminished, local agriculture would certainly suffer. The effects on the rest of the community of Balmorhea would be devastating since they depend on the aquifer and the spring flows for everything from domestic water to tourism.

Although current state law would allow unrestricted pumping from the aquifer that supports the Balmorhea springs complex, one thing that can prevent overpumping is the Federal Endangered Species Act. The Endangered Species Act protects the fish, the fish need the water, and as long as the water is flowing from the springs it also is available to humans. Through a pragmatic understanding of the basic relationship between the natural and human communities, biologists and Balmorhea community leaders chose to work together on a solution that would benefit all concerned rather than adopt adversarial roles. While the farmers had previously viewed the fish as something that hampered and perhaps threatened their livelihood, the fish may actually be their best insurance for sustained spring flows.

A plan was formulated to create a ciénega to look and function like a natural ecosystem. RCWID and the agricultural community it represents agreed to provide the essential water needed to create a secure environment for the endangered fishes. Water is a rare and precious commodity in West Texas, particularly for farmers, but by each of the users giving up a small amount, they would be providing insurance for future water supplies.

An additional benefit for the farmers was that, because of their help in creating preferred habitat for

the endangered fishes, the Texas Department of Agriculture (TDA), the U.S. Fish and Wildlife Service (USFWS) and the U.S. Environmental Protection Agency (EPA) proposed a plan to allow the benefits of the *ciénega* to offset any potential effects from pesticide use on farms that could impact the endangered species in the irrigation canals. The fish would have a better place to live and the farmers could continue to raise their crops.

Biologists, engineers and resource managers from universities and government agencies joined forces to make the project work. The U.S.D.A. Natural Resource Conservation Service provided soil analysis and, along with staff from the Texas Agricultural Extension Service, TDA, the University of Texas at Austin and the University of Texas-Pan American, gave expert advice on some of the intricacies of the project. The expertise of the Texas Department of Transportation also was crucial. Their surveyors, design engineers and equipment operators transformed biological concepts into reality. The Texas Department of Criminal Justice provided inmates to build the Observation Deck and retaining walls as well as install the plant materials selected for the initial *ciénega* vegetation restoration. Botanists at Sul Ross State University provided container-grown native plants for the project. The one-of-a-kind window wall was designed, built, transported and installed by a beneficent concrete fabrication company located 500 km away.

Funding for the San Solomon *Ciénega* was provided by grants from the Educational Foundation of America and the National Fish & Wildlife Foundation. Additionally, fabrication costs for the window wall were provided by a TDA grant from the EPA and with contributions from the Texas Organization for Endangered Species.

In 1996, construction of the 1-ha San Solomon *Ciénega* was completed. This wetland is situated on Balmorhea State Park land within the boundaries of the original, natural *ciénega*. As a result, the native fish fauna, including Comanche Springs pupfish and Pecos gambusia, has flourished. This location now provides a natural habitat and contains the largest known concentration of Comanche Springs pupfish. Recent monitoring efforts have resulted in average estimates of the summer population of pupfish in the *ciénega* at 270,000 individuals.

Aquatic plants indigenous to *ciénegas*, as well as grasses and shrubs characteristic of the drier aspects of these desert wetland communities, were planted in the *ciénega* and now are well established. Some of the more common species are common cattail (*Typha latifolia*), common reed (*Phragmites australis*), alkali bulrush (*Scirpus maritimus*), hardstem bulrush (*S. acutus*), Olney bulrush (*Scirpus olneyi*), sand spikerush (*Eleocharis montevidensis*), buttonbush (*Cephalanthus occidentalis*), Goodding willow (*Salix gooddingii*), Rio Grande cottonwood (*Populus deltoides* var. *wislizenii*), four-winged saltbush (*Atriplex canescens*), alkalai sacaton (*Sporobolus airoides*), big sacaton (*Sporobolus wrightii*), tobosa (*Hilaria mutica*), granjeno (*Celtis pallida*) and western soapberry (*Sapindus saponaria* var. *drummondii*). Beyond the immediate wetted perimeter of the *ciénega*, the habitat grades into a desert plains grassland that was once common to the region.

Many species of birds, reptiles and mammals began to use the new wetland almost immediately. These include belted kingfisher (*Megaceryle alcyon*), black phoebe (*Sayornis nigricans*), swallows (*Petrochelidon* spp.), white-throated swift (*Aeronautes saxatalis*), green heron (*Butorides virescens*), swamp sparrow (*Melospiza georgiana*), yellow-headed blackbird (*Xanthocephalus xanthocephalus*), sora (*Porzana carolina*), yellowthroat (*Geothlypis trichas*), blotched watersnake (*Nerodia erythrogaster transversa*), spiny softshell turtle (*Trionyx spiniferus*), pond slider (*Chrysemys scripta*), javelina (*Pecari angulatus*) and desert cottontail (*Sylvilagus auduboni*).

People of the local and regional community and state park visitors benefit from a living exhibit that shows the importance of the springs and their wetlands for the fishes and other wildlife of West Texas. Because the primary purpose of the *ciénega* is to provide desert wetland habitat, visitor access is limited to only a small portion of the total restoration. However, TPW has tried to maximize the aesthetic and educational experiences available at locations that are accessible to the public. The observation deck provides an unobstructed view of most of the above-water portion of the *ciénega*, and the clear water allows viewing of much of its underwater life. The window wall was custom designed for San Solomon *Ciénega* so that visitors would have a view that few have seen—life in the *ciénega* as its aquatic residents see it.

Another attempt to further protect Comanche Springs pupfish was not so successful. Lake Balmorhea is a downstream, 200-ha storage reservoir for irrigation water from the Balmorhea springs complex and contained an introduced population of sheepshead minnow (*C. variegatus*). This species is known to compete and hybridize with Comanche Springs pupfish (Stevenson and Buchanan, 1973; Echelle and Echelle, 1994). In 1998, the RCWID allowed TPW to partially drain the reservoir and attempt to remove all fish by application of rotenone, a fish toxin. The objective of the project was to eliminate sheepshead minnow from the reservoir in order to remove the threat of hybridization with Comanche Springs pupfish. A large population of sheepshead minnow inhabited the lake; post-rotenone extrapolation of subsamples put the estimate at 5,000,000. Unfortunately and for undetermined reasons, some sheepshead minnow survived the rotenone treatment. They have since begun repopulating the reservoir. Part of the agreement with RCWID was to restock the reservoir with sport fishes in order to improve tourism in the area. These

piscivores should help keep the numbers of sheepshead minnow in check.

The creation of the San Solomon Ciénega was accomplished through willing participation of diverse entities with a common goal of mutual benefit. During the last decade, USFWS developed a formal method of participation in such projects (for non-listed species) through Conservation Agreements. The successful cooperation among private and government entities in the San Solomon Ciénega project served as a precursor to future conservation efforts by TPW. Conservation Agreements are relatively new and some have not worked. Therefore, they are closely scrutinized by not only the USFWS, but also by others from across the political spectrum. In order to be effective, a Conservation Agreement must provide some immediate reduction in threat to the species and provide long-term security against extinction. When well designed, they provide a format and incentives for affected entities to work together to resolve issues and conserve natural resources.

DEVILS RIVER MINNOW CONSERVATION AGREEMENT

A Conservation Agreement among the City of Del Rio, TPW and USFWS was implemented in 1998. Due to the cooperative efforts outlined in the Agreement, Devils River minnow (*Dionda diaboli*) was listed as threatened rather than endangered (USFWS, 1999). The Agreement details a 5-year plan of research and conservation actions that are designed to resolve the threats to the Devils River minnow and lead to its ultimate de-listing. Benefits include protection of water quality and quantity in the Devils River and adjacent streams for both fish and people and creation of a green belt/stream corridor along San Felipe Creek in the City of Del Rio that not only provides quality habitat for fishes, but will also provide a nature-friendly, city park and potential for increased tourism.

The Devils River (Figure 2) is one of the most pristine rivers in southwestern North America. Due to its geographic location and historic stability, the Devils River sustains many indigenous organisms. It remains relatively unpolluted and undammed and although spring flows have diminished, they are still substantial. Lim-

ited access has kept the river from being thoroughly studied by the scientific community; however, collections in the past decade by Garrett et al. (1992) and others indicate a diminution in abundance of most flowing-water species, particularly Devils River minnow. A survey in 1953 showed Devils River minnow was the fifth-most abundant fish species at Baker's Crossing and the sixth-most abundant fish in the upper Devils River (Hubbs and Brown, 1956). In the mid-1970s, Harrell (1978) found it remained the sixth-most abundant fish (in 72 collections, Harrell averaged 24-25 *D. diaboli* per collection). In 1988-1989, collections from 25 locations throughout the historic range in the United States yielded a total of only 7 individuals: Devils River = 2; San Felipe Creek = 3; Sycamore Creek = 2 (Garrett et al., 1992). The numbers had declined such that it was rare where it occurred at all and was probably the least abundant of the approximately 30 species that occur in these streams. In 1979, Devils River minnow made up 6-18% of the *Dionda* population at the Head Spring area of San Felipe Creek. In 1989, none were present there (Garrett et al., 1992).

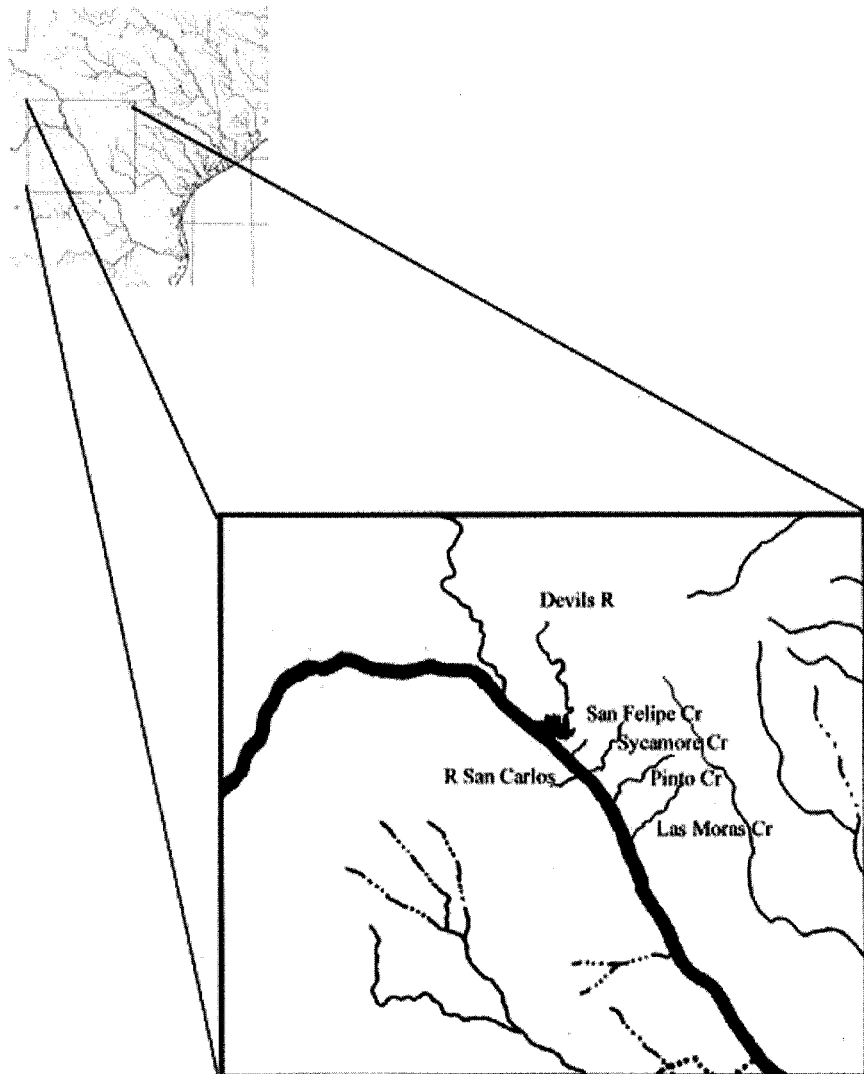


Figure 2. Range map for Devils River minnow.

Members of the genus *Dionda* are specialized for living in spring-fed, flowing waters and are found primarily in Texas and Mexico (Garrett et al., 1992). Devils River minnow is distinguished from other species of the genus by a variety of characters, including distinctive color pattern, narrow head and number of lateral line scales (Hubbs and Brown, 1956). Biochemical work in the past decade has further distinguished unique characteristics of this species (Gold et al., 1992; Mayden et al., 1992).

The Devils River minnow is known to occur in the Devils River, San Felipe Creek and Sycamore Creek, Val Verde County, Texas. It historically occurred in Las Moras Creek, Kinney County, Texas, but was eliminated from that locality sometime before 1980 (Smith and Miller, 1986; Garrett et al., 1992). Extirpation was likely due to periodic failure of the springs from drought and groundwater pumping and from modifications to the spring for construction and maintenance of a swimming pool. There are also historic records of occurrence in two small streams in Coahuila, Mexico, the Río San Carlos and Río Sabinas (Contreras-B. and Lozano-V., 1994). Their current status there is unknown; no collection attempts have been made since the early 1970s.

Although very little is known of the ecology of the Devils River minnow, some threats are apparent. Range reduction has occurred by extirpation of the Las Moras Creek population, minimal flows in Sycamore Creek and inundation of the lower Devils River first by Walk and Devils lakes earlier in the 20th century and ultimately Amistad Reservoir in 1968. Many springs in the area have diminished flows and some have totally stopped (e.g., Beaver Springs, Juno Springs and Dead Man's Hole), thus reducing the overall length of the Devils River as well as the quantity of water flowing in it (Brune, 1981). Many of the area's perennial streams, listed by Gray (1919), no longer flow. In the Devils River, U. S. Geological Service data from the Pafford's Crossing gauging station reveals a general decrease in daily mean discharge for the period between the study by Harrell (1978) and that of Garrett et al. (1992).

The Devils River minnow may be suffering from biological threats as well. Numerous exotics have become established in the area, including common carp

(*Cyprinus carpio*), Gulf killifish (*Fundulus grandis*), redbreast sunfish (*Lepomis auritus*), smallmouth bass (*Micropterus dolomieu*) and blue tilapia (*Oreochromis aureus*). Although fishes throughout the Chihuahuan Desert have been negatively impacted by introduced species (Hubbs, 1990) and such factors as predation and competition may be causing negative impacts on the native fish fauna, specific effects on Devils River minnow are not known. Experiments designed to elucidate these interactions are ongoing.

Much of the water for San Felipe Creek comes from two large springs (San Felipe Springs) within the City of Del Rio. The City also gets its municipal water supply from San Felipe Springs. Conserving the quantity and quality of water from the springs is critical for both the Devils River minnow and the citizens of Del Rio.

The USFWS proposed the Devils River minnow for listing as threatened in 1978 with critical habitat proposed for portions of San Felipe Creek and the Devils River. The USFWS withdrew the proposal in 1980 and retained its designation as a candidate species. The USFWS published a new proposal to list the Devils River minnow as endangered in March, 1998.

During the period 1997-1998, TPW was working with USFWS, Del Rio and private landowners to develop ways to protect the minnow. Landowners and city officials feared repercussions of listing the fish as endangered and came to understand that a cooperative approach to restoring and protecting the ecosystem would be the best for all concerned. The Devils River Minnow Conservation Agreement was signed by the USFWS, TPW and the City of Del Rio in September 1998. The TPW worked closely with city officials and local landowners to develop conservation actions that were beneficial to the species. Those actions in the Agreement include determining the current status of the species throughout its range, maintaining captive populations for reintroductions in nature, protecting the San Felipe Creek watershed, providing technical assistance to landowners on riparian protection and management, revising live bait harvest and selling practices in the Devils River area to prevent the further establishment of exotic, aquatic species and additional ecological research, including interactions between Devils River minnow and smallmouth bass.

The Conservation Agreement provides a positive incentive for cooperative actions by all parties and yields scientific access to previously unavailable locations. The primary motivation for the Conservation Agreement was to remove the threats to the Devils River minnow sufficiently so that protection under Federal law was not necessary. The USFWS carefully considered the Agreement and to what extent it had been

implemented as of the time the listing decision was due. The USFWS concluded that with an accelerated implementation schedule, a listing determination of threatened rather than endangered would be appropriate. If successful, the Devils River minnow and other aquatic fauna will be protected and the quantity and quality of streams throughout the range will also be insured.

PECOS PUFFISH CONSERVATION AGREEMENT

The Pecos Pupfish Conservation Agreement was initiated in 1999. The parties involved are TPW, New Mexico Department of Game and Fish, New Mexico Department of Agriculture, New Mexico Division of State Parks, Bureau of Land Management (BLM) and USFWS. The Agreement provides conservation measures, new bait fish regulations and creation of additional habitat. It is designed to preclude the need to list the Pecos pupfish (*Cyprinodon pecosensis*) as a federally endangered species by reducing threats to the species and establishing populations in newly created habitats adjacent to the Pecos River in Texas. This effort incorporates the help of private landowners through the federally funded, state administered, Landowner Incentive Program. With financial assistance and biological guidance provided by TPW, landowners have modified private ponds to mimic natural desert wetland habitat while still allowing the landowner's original intentions for the water body. Due to the ability of the Agreement to remove immediate threats to the fish, a proposal to list it as endangered was withdrawn by USFWS in 2000.

The Pecos pupfish is endemic to the Pecos River system (Figure 1) from the vicinity of Roswell, New Mexico to the mouth of Independence Creek, Terrell County, Texas (Echelle and Echelle, 1978). It now only occurs in Salt Creek, a small Pecos River tributary in Texas, and at Bitter Lake National Wildlife Refuge and Bottomless Lakes State Park in New Mexico (Hoagstrom and Brooks, 1999; Echelle et al., this volume). It also occurs sporadically in the Pecos River upstream of Artesia, New Mexico (Propst, 1999).

Pecos pupfish can occur in a variety of habitats and water qualities. It can flourish in locations that fluctuate in water quantity and chemistry, ranging from

highly saline sinkholes to typical desert streams. It has been found in locations with dissolved chlorides ranging from 185 mg/l to 8,940 mg/l (Davis, 1981). Age at reproductive maturity, ovary size, egg size and egg number vary among populations, and are apparently associated with population density (Garrett, 1982). The distribution is mostly limited by interspecific interactions and thus, it is typically found in habitats that are low in species diversity (Echelle and Echelle, 1978).

During 1954, Pecos pupfish was the most abundant fish in the Pecos River between New Mexico and Sheffield, Texas (Echelle et al., 1997). Abundance has declined dramatically since the early 1980s, when non-native sheepshead minnow was introduced into the Pecos River in Texas. The original introduction appears to have been a baitfish release in Red Bluff Reservoir (Childs et al., 1996). As a result, Pecos pupfish was eliminated from the lower Pecos River upstream to Loving, New Mexico, and replaced with a hybrid swarm (Echelle and Conner, 1989; Wilde and Echelle, 1992; Echelle et al., 1997). Pecos pupfish is listed as "threatened" by Texas and New Mexico and is considered a "species of concern" by the American Fisheries Society (Williams et al., 1989). Their status is due to habitat loss and especially to hybridization with sheepshead minnow.

The Texas portion of the Conservation Agreement consists of establishing off-channel populations near the Pecos River through a cooperative program with private landowners, developing more restrictive bait-fish regulations, attempting to remove sources of sheepshead minnows from West Texas and monitoring the status of Pecos pupfish in Texas. Progress has been made in all categories.

New Mexico's commitment is similar to Texas, with the addition of ensuring the security of populations on state park lands. The USFWS and BLM are also protecting populations on federal lands as well as providing some funding.

This Conservation Agreement is the most feasible approach to addressing the biological problems of this fish. Although restrictive rules inherent in endangered species status are often necessary to prevent extinction, in the case of the Pecos pupfish, the opposite is true. Creation of new habitat on private lands would likely not be possible if there was a poten-

tial for negative impacts on private enterprise. With the Conservation Agreement, landowners are willing to have Pecos pupfish stocked in their ponds and federal grants can be used to help develop suitable habitat in the ponds. To date, two shrimp farmers, using naturally saline groundwater in West Texas, have joined the program. At each location, a *ciénega* was constructed (1.7 ha and 7 ha) to provide secure habitat and Pecos pupfish from Salt Creek were stocked in 2000 and 2001. The program allows and encourages individuals to participate in the conservation of rare, natural resources without personal risk or liability.

SUMMARY

In each of the above projects, many of the critical conservation actions would not have been possible without the cooperation of public and private entities. These are examples of a positive approach to problem

resolution that provides benefits to each of the cooperators. These "win-win" situations are not always available, but are certainly the most desirable.

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