

Chapter 5:

Fishes of Southwestern Grasslands: Ecology, Conservation, and Management

Introduction

The world possesses an amazing number and variety of freshwater fishes. Fish, the most diverse taxon on Earth, exhibit more species than all vertebrate taxa combined (Hocutt and Wiley 1986, May 1988, Nelson 1994, Matthews 1998) (table 5-1). Worldwide, an estimated 28,500 fish species are in 57 orders and 482 families (Nelson 1994). The North American freshwater fish fauna is also relatively rich and highly diverse ranging from primitive forms such as the *Petromyzontidae* (lampreys) to more modern and advanced forms such as the *Percidae* (perches) (Miller 1959, Gilbert 1976, Briggs 1986, Cavender 1986, Nelson 1994). Currently, there are about 1,000 known freshwater fish species representing 50 families in 201 genera in an area from Canada and Alaska south to the Isthmus of Tehuantepec (Gilbert 1976, Lee and others 1980, Briggs 1986, Mayden 1992a,b). Some of these fish families are endemic to North America whereas others are also found on bordering continents (Briggs 1986). Despite persistent research for more than 100 years, numbers of described fishes in North America continue to increase over time. For example, Lee and others (1980) described 770 North American fishes, whereas just over a decade later Mayden (1992a,b) listed 971.

Regionally significant differences in North American freshwater fish species numbers and diversity are evident. The fish fauna east of the Continental Divide,

consisting of eight speciose genera that comprise the majority of freshwater fish species on the continent, is species rich compared to the fish fauna of the Southwestern grasslands of New Mexico, Texas, and Oklahoma and the fish fauna west of the Continental Divide (Smith 1981b, Briggs 1986, Minckley and others 1986, Moyle and Herbold 1987, Etnier and Starnes 1993). Concentrations or “hot spots” of fish species diversity east of the Divide occur in the Appalachian and Ozark highlands and in Eastern streams. Here, some river systems contain as many as 100 to 200 fish species (Robison and Beedles 1974, Hocutt and others 1986, Schmidt 1986, Starnes and Etnier 1986, Mayden 1987, Robison and Buchanan 1988, Etnier and Starnes 1993, Matthews 1998). Diversity of fishes is also high throughout the Mississippi-Missouri River drainage, the region’s major basin, where at least 260

Table 5-1. Fish versus tetrapod diversity. Numbers given are for described species worldwide in each taxon. (Adapted from Nelson 1994 and May 1988)

Species	Numbers
Fishes	28,618
Amphibians	2,800
Reptiles	6,000
Birds	4,500
Mammals	4,500

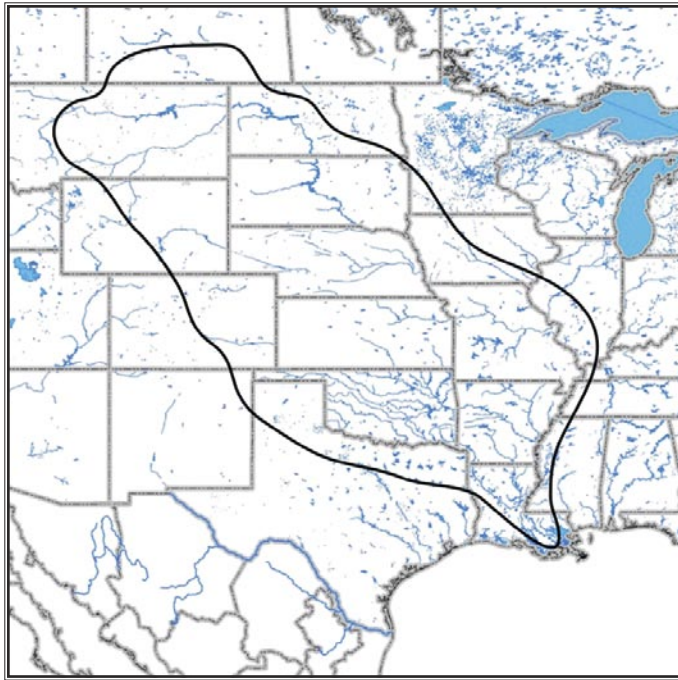


Figure 5-1. Western Mississippi Basin.

species of fresh water fishes are found (Cross 1967, Smith 1981b, Cross and others 1986, Robison 1986, Moyle and Herbold 1987). Moving west into the western Mississippi Basin (fig. 5-1) the number of native fish species decline to about 235 and continues to decline as one moves westward into Western short-grass prairie and Southwestern grasslands east of the Continental Divide (fig. 5-2).

In contrast, the fish fauna west of the Continental Divide is characterized by relatively depauperate fish assemblages rich in endemic species, many of which are confined to a single spring, stream, or drainage (Minckley 1973, Pister 1974, Smith 1981a,b, Briggs

1986, Minckley and others 1986, Moyle and Herbold 1987). The fauna west of the Divide has only about one-fourth as many species as the waters of Eastern North America (Miller 1959, Briggs 1986, Minckley and others 1986). For example, Burr and Mayden (1992) suggest that the Colorado River basin has approximately 32 native fish species, whereas 375 native fishes could be found in the Mississippi River. Matthews and Gelwick (1990) indicated that a typical reach (200 m) in a medium or large stream in Eastern North America contained 20 species of fish, whereas a reach in semiarid or arid lands of the Midwestern to Western North America may only contain eight to 12 species. Moyle and Herbold (1987) suggested that a sample of fishes in Western North America could contain fewer than 10 species, whereas in the Mississippi River basin two to three times as many fishes could be found.

The variation in fish species diversity and numbers between the Mississippi Basin and Southwestern grasslands east of the Continental Divide and the fish fauna west of the Continental Divide can be linked to a myriad of obvious and less apparent factors such as historic and recent evolutionary geology, effects of inland seas, climate, past and present erosive processes, and extant local fish faunas (Endler 1977, Briggs 1986, Minckley and others 1986, Smith and Miller 1986). In general, the fish fauna east of the Divide experienced a relatively calm geologic history. During the Pleistocene, the fish fauna of the East was able to avoid a high number of extinctions due to the region's large watershed size, stability, and lack of barriers to fish migration to refugia during glaciation and subsequent recolonization of habitats as ice sheets retreated northward (Smith 1981). In contrast, the fish fauna of the eastern Rocky Mountain slopes and west of the Continental Divide experienced more intense geological events during their evolution. West

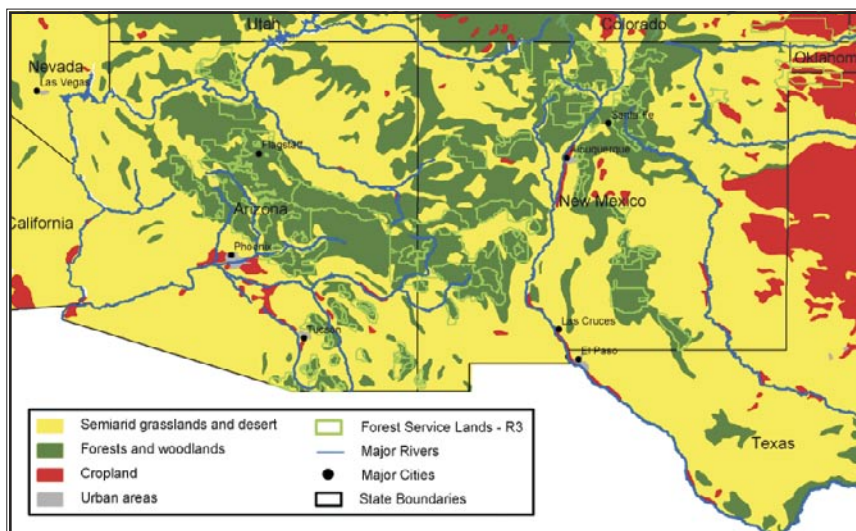


Figure 5-2. Semiarid grasslands, deserts, and major rivers of the Southwestern United States.

of the Divide, the fish fauna evolved in small isolated basins that were influenced by periods of extreme tectonic uplift, regional erosion, mountain building, volcanism, and increasing aridity (Dickinson 1981, Minckley and others 1986). In contrast to events east of the Continental Divide, these processes created barriers to fishes moving in search of refugia and prevented dispersal and recolonization of waters subsequent to these geologic events (Smith 1981b, Dott and Batten 1981). Isolation, fragmentation, and allopatric speciation prevailed in this region. Many fish populations were extirpated, others were isolated or restricted to small habitats where genetic variation of the population was limited and speciation was allowed to proceed (Miller 1946a,b, 1959, Simberloff 1966, Moyle 1976, Smith 1978, 1981a,b, Minckley and others 1986). These factors, coupled with loss of genetic variation, contributed to rapid evolutionary change (Hubbs and Miller 1948, Hubbs and others 1974). These events account for a fish fauna that is characterized by special adaptations, endemism, and relatively depauperate populations.

Population Trends

As illustrated, the Southwestern fish fauna evolved in a region with highly variable climate and a dynamic geologic history. These environmental influences are reflected in the specializations and unique adaptations of the native Southwestern fish fauna. Unfortunately, the adaptations that have allowed these fishes to survive the harsh hydrologic conditions of the Southwest uplands and grasslands also made them vulnerable to large-scale human-induced ecological changes (Johnson and Rinne 1982, Minckley and Deacon 1991).

Native fishes of the Southwestern United States have experienced a severe reduction in both range and numbers since the region was first surveyed in the 19th century (Williams and others 1985). Significant fish faunal changes were first observed in the early 1900s (Miller 1946a, 1961, Minckley and Deacon 1968, Minckley and Deacon 1991, Koster 1957, Tyus and others 1982, Sublette and others 1990). By this time, extirpation and decline of native forms due to habitat alteration and competition with nonnative fish species had already begun (Miller 1961, Scurlock 1998). As early as 1904, nonnative common carp (*Cyprinus carpio*), brown bullhead (*Ameiurus nebulosus*), mosquito fish (*Gambusia affinis*), and green sunfish (*Lepomis cyanellus*) were collected in the Gila River of Arizona and New Mexico (Miller and others 1991). By 1926, 50 percent of the native fish fauna had been extirpated in the Salt River near Tempe, AZ, and by the late 1950s only two (14 percent) of the 14 original native fishes remained (Minckley and Deacon 1968). By the turn of the century, four fish species inhabiting grassland stream reaches

including shovelnose sturgeon (*Scaphirhynchus platyrhynchus*), spotted gar (*Lepisosteus osculatus*), bonytail chub (*Gila elegans*), and freshwater drum (*Aplodinotus grunniens*), had been extirpated in New Mexico (Koster 1957, Sublette and others 1990). Noticing this downward trend prompted Miller (1946b) to call for studies of Western fishes to ensure their persistence.

It was not until the middle of the 20th century, however, when large-scale continental and regional changes in the native fish faunas occurred and were finally recognized by the scientific community (Miller 1961, 1972, Minckley and Deacon 1991). Miller (1961) examined and provided data on changes in the Southwestern fish fauna. In a more extensive effort, Miller (1972) provided a preliminary effort to identify threatened fishes of the 50 States and again highlighted the decline of Southwestern native fishes, provided reasons for the decline, and listed species of threatened fishes for each of the 50 States in the United States (Miller 1972). He listed 305 fishes as threatened. Nine fish species were listed as threatened in Arizona and five in New Mexico. Miller and others (1989) documented the extinction of three genera, 27 species, and 13 subspecies of fishes from North America during the past 100 years.

Today, most native Western fishes have become listed as threatened or endangered by the USDI Fish and Wildlife Service, and others are given protection afforded by individual States (Johnson and Rinne 1982, Williams and others 1989, Minckley and Douglas 1991, USFWS 2003). Of the 150 fishes recognized by Lee and others (1980) west of the Continental Divide, 122 are considered to be in danger of extirpation (Minckley and Deacon 1991). A large percentage (34 percent) of the fishes currently listed by the USDI Fish and Wildlife Service are from the Western United States (USFWS TESS 2003). Viability of many of these populations is questionable. Many species not afforded protection under the Endangered Species Act or by State regulations are also believed to be in decline.

In New Mexico, a minimum of 66 and possibly as many as 70 native fishes were extant when Euro-American settlers arrived (approximately 1525) (Sublette and others 1990, Propst 1999). Using the latter value, since that time, two (3 percent) native fish species have gone extinct, 12 (17 percent) have been extirpated from the State, 25 (36 percent) are considered to be in decline, and 31 (44 percent) are listed as stable (table 5-2). Of the remaining native fish fauna, 23 (41 percent) are listed as threatened or endangered by the State of New Mexico (Propst 1999). Of these, 10 (18 percent) are listed as threatened and endangered by the USDI Fish and Wildlife Service, and one additional species is proposed for listing (table 5-2). Many of these are species found in reaches that flowed through grasslands of the Southwest (Koster 1957, Sublette and others 1990, Propst 1999).

Table 5-2. Status of native fishes in New Mexico.

Species Scientific name	Common name	Status	
		USFWS	NM
<i>Scaphirhynchus platyrhynchus</i>	Shovelnose sturgeon		Ex
<i>Lepisosteus oculatus</i>	Spotted gar		Ex
<i>Lepisosteus osseus</i>	Longnose gar		
<i>Anguila rostrata</i>	American eel		
<i>Dorosoma cepedianum</i>	Gizzard shad		
<i>Oncorhynchus clarki virginalis</i>	Rio Grande cutthroat trout		
<i>Oncorhynchus gilae</i>	Gila trout	En	Th
<i>Oncorhynchus clarki</i>	Colorado River cutthroat		Ex
<i>Astyanax mexicanus</i>	Mexican tetra		Th
<i>Agosia chrysogaster</i>	Longfin dace		
<i>Camptostoma anomalum</i>	Central stoneroller		
<i>Cyprinella Formosa</i>	Beautiful shiner		Ext
<i>Cyprinella lutrensis</i>	Red shiner		
<i>Dionda episcopa</i>	Roundnose minnow		
<i>Extrarius aestivalis</i>	Speckled chub		
<i>Macrhybopsis aestivalis tetranemus</i>	Arkansas River speckled chub		Th
<i>Gila elegans</i>	Bonytail chub		Ex
<i>Gila nigrescens</i>	Chihuahua chub		En
<i>Gila Pandora</i>	Rio Grande chub		
<i>Gila nigrescens</i>	Chihuahua chub	Th	En
<i>Gila intermedia</i>	Gila chub		Ex
<i>Gila robusta</i>	Roundtail chub		En
<i>Hybognathus amarus</i>	Rio Grande silvery minnow	En	En
<i>Hybognathus placitus</i>	Plains minnow		
<i>Meda fulgida</i>	Spikedace	Th	Th
<i>Notropis amabilis</i>	Texas shiner		Ex
<i>Notropis girardi</i>	Arkansas shiner	Th	En
<i>Notropis jemezianus</i>	Rio Grande shiner		
<i>Notropis orca</i>	Phantom shiner	Ext	Ext
<i>Notropis simus simus</i>	Bluntnose shiner	Ext	Ext
<i>Notropis simus pecosensis</i>	Pecos Bluntnose shiner		Th
<i>Notropis stramineus</i>	Sand shiner	Th	
<i>Phenacobius mirabilis</i>	suckermouth minnow		Th
<i>Phoxinus erythrogaster</i>	Southern redbelly dace		En
<i>Pimephales promelas</i>	Fathead minnow		
<i>Platygobio gracilis</i>	Flathead chub		
<i>Ptychocheilus lucius</i>	Colorado cikeminnow	En	En
<i>Rhinichthys cataractae</i>	Longnose dace		
<i>Rhinichthys osculus</i>	Speckled dace		
<i>Semotilus atromaculatus</i>	Creek chub		
<i>Tiaroga cobitis</i>	Loach minnow	Th	Th
<i>Carpoides carpio</i>	River carpsucker		
<i>Catostomus commersoni</i>	White sucker		
<i>Catostomus plebeius</i>	Rio Grande sucker		
<i>Catostomus clarki</i>	Desert sucker		
<i>Catostomus discobolus</i>	Zuni bluehead sucker		En
<i>Catostomus insignis</i>	Sonoran sucker		
<i>Catostomus latipinnis</i>	Flannelmouth sucker		
<i>Cycleptus elongatus</i>	Blue sucker		En
<i>Xyrauchen texanus</i>	Razorback sucker		
<i>Ictiobus bubalus</i>	Smallmouth buffalo		Th
<i>Moxostoma congestum</i>	Gray redbreast		
<i>Ictalurus furcatus</i>	Blue catfish		
<i>Ictalurus lupus</i>	Headwater catfish		
<i>Ictalurus melas</i>	Black bullhead		
<i>Ictalurus punctatus</i>	Channel catfish		
<i>Pylodictus olivaris</i>	Flathead catfish		

Table 5-2. Continued.

Species Scientific name	Common name	Status	
		USFWS	NM
<i>Cyprinodon pecosensis</i>	Pecos pupfish	Th	Th
<i>Cyprinodon tularosa</i>	White sands pupfish		Th
<i>Cyprinodon sp.</i>	Palomas pupfish	Ext	Ext
<i>Fundulus zebrinus</i>	Plains killifish		
<i>Lucania parva</i>	Rainwater killifish		
<i>Poeciliopsis occidentalis</i>	Gila topminnow	En	Th
<i>Gambusia affinis</i>	Mosquito fish		
<i>Gambusia noblis</i>	Pecos gambusia	En	En
<i>Lepomis cyanellus</i>	Green sunfish		
<i>Lepomis macrochirus</i>	Blugill		
<i>Lepomis megalotis</i>	Longear sunfish		
<i>Micropterus salmoides</i>	Largemouth bass		
<i>Etheostoma lepidum</i>	Greenthroat darter		Th
<i>Percina macrolepida</i>	Bigscale logperch		Th
<i>Aplodinotus grunniens</i>	Freshwater drum		Ex
<i>Cottus bairdi</i>	Mottled sculpin		

En=Endangered, Th =Threatened, Ext = Extinct, Ex = Extirpated (For more detailed description of Federal and State threatened and endangered status of native fishes see Propst 1999.)

Similar to New Mexico, the native fish fauna of Arizona has been greatly modified over the past 120 years. Of the 36 fishes native to the waters of Arizona, one (3 percent) is now extinct, three (8 percent) have been extirpated from the State, 21 (58 percent) are in decline, and 11 (25 percent) are considered stable (table 5-3). Of the remaining native fishes, 25 (69 percent) are listed as Wildlife of Special Concern in Arizona, 19 (53 percent) are listed as threatened or endangered by the USDI Fish and Wildlife Service, and one is classified as a candidate for listing (table 5-3).

The decline of native fishes and native fish communities can be linked to resource extraction, alteration of stream channels, and water diversion by indigenous peoples and Euro-Americans settling in the region (approximately 1521) (Minckley and Douglas 1991, Rinne and Platania 1995, Calamusso and Rinne 1999). Introduction of nonnative aquatic species and associated competition and hybridization have resulted in the decline, alteration, and extirpation of many native fish species and entire assemblages. Remaining populations of native fish are often isolated, then decline to low levels leading to categorization as threatened, endangered, or at risk (Miller 1961, Deacon and others 1979, Williams and Sada 1985, Williams and others 1985, 1989, Miller and others 1989, Rinne and Minckley 1991). Often resource management agencies do not have complete knowledge of populations that in fact warrant listing or some type of protection.

Biological Threats: Introduction of Nonnative Fish Species

Other than dewatering of a stream reach or desiccation of some type of standing body of water the primary threat to the persistence of native fishes in Southwestern grasslands has been the introduction and range expansion of nonnative fishes. In fact, Minckley and Douglas (1991) suggested that the spread of nonnative fishes may be more detrimental than all other environmental perturbations combined. The effects of dewatering, habitat loss, and pollution are sometimes temporary, but once a nonnative species is established it is difficult or impossible to remove it from many systems (Rinne and Turner 1991).

Nonnative fishes have been introduced into the waters of North America since the late 1600s. Fuller and others (1999) listed 536 fish taxa (representing 75 families) that have been transplanted into the United States outside their native range. Unfortunately, rates of introduction have continued to increase over the past 45 years due to more efficient transportation and demand for nonnative forms for sport, fish farming, bait, and aquarium use (Fuller and others 1999, Heidinger 1999, Li and Moyle 1999, Nico and Fuller 1999). Of the myriad of intentional fish stockings, only a few have been positive, and no unplanned introductions have been considered beneficial (Steirer 1992, Courtney and Williams 1992). Introductions have, on the whole, caused many problems for native fishes (Deacon 1979, Deacon and others 1979, Williams and

Table 5-3. Status of native fishes in Arizona.

Species		Status	
		USFWS	AZ
<i>Elops affinis</i>	Machete		
<i>Mugil cephalus</i>	Striped mullet		
<i>Oncorhynchus gilae apache</i>	Apache trout	Th	X
<i>Oncorhynchus gilae gilae</i>	Gila trout	En	X
<i>Campostoma ornatum</i>	Mexican stoneroller		
<i>Cyprinella formosa</i>	Beautiful shiner	Th	X
<i>Gila cypha</i>	Humpback chub	En	X
<i>Gila ditaenia</i>	Sonora chub	Th	X
<i>Gila elegans</i>	Bonytail chub	En	X
<i>Gila intermedia</i>	Gila chub	C	X
<i>Gila nigra</i>	Headwater chub		
<i>Gila purpurea</i>	Yaqui chub	En	X
<i>Gila robusta</i>	Roundtail chub		X
<i>Gila seminuda</i>	Virgin chub	En	X
<i>Meda fulgida</i>	Spikedace	Th	X
<i>Lepidomeda mollispinis mollispinis</i>	Virgin spinedace		X
<i>Lepidomeda vittata</i>	Little Colorado spinedace	Th	X
<i>Plagopterus argentissimus</i>	Woundfin	En	X
<i>Ptychocheilus lucius</i>	Colorado sikeminnow	En	X
<i>Rhinichthys osculus</i>	Speckled dace		
<i>Tiaroga cobitis</i>	Loach minnow	Th	X
<i>Catostomus bernardini</i>	Yaqui sucker		X
<i>Catostomus clarki</i>	Desert sucker		
<i>Catostomus discobolus</i>	Bluehead sucker		
<i>Catostomus d. yarrowi</i>	Zuni Mountain sucker		
<i>Catostomus insignis</i>	Sonora sucker		
<i>Catostomus latipinnis</i>	Flannelmouth sucker		X
<i>Catostomus sp.</i>	Little Colorado sucker		
<i>Xyrauchen texanus</i>	Razorback sucker	En	X
<i>Ictalurus pricei</i>	Yaqui catfish	Th	X
<i>Cyprinodon eremus</i>	Quitobaquito pupfish	En	X
<i>Cyprinodon macularius</i>	Desert pupfish	En	X
<i>Cyprinodon sp.</i>	Monkey spring pupfish	Ext	Ext
<i>Poeciliopsis occidentalis occidentalis</i>	Gila topminnow	En	X
<i>Poeciliopsis occidentalis sonoriensis</i>	Yaqui topminnow	En	X

En = Endangered, Th = Threatened, Ext = Extinct, X = Species of Special Concern by State of Arizona, Ex = Extirpated

others 1989). The nonnative fishes currently found in almost all major watersheds of the Continental United States have led to the decline and listing of native fish populations across North America (Taylor and others 1984, Courtenay and others 1984, Courtenay and Stauffer 1990, Hendrickson and Brooks 1991, U.S. Congress, OTA 1993). Today, 70 percent of the fishes listed under the Endangered Species Act have been negatively impacted by nonnative fishes (Lassuy 1995, USFWS 2003, Tyus and Saunders 2000). Further, it is estimated that the extinction of 20 native fishes in the 1900s is primarily due to interactions with nonnative forms (Miller and others 1989).

Impacts of nonnative fish on native fishes/communities depend on the physiological, behavioral, and ecological capabilities of the nonnative species and the

biological and physical components of the ecosystem and its inhabitants (Taylor and others 1984, Haines and Tyus 1990, Rinne and others 1998). Nonnative fish often have adaptations that provide advantages in the aquatic environment in which they are introduced. These capabilities (reproductive, environmental tolerances, behavior) enable them to exploit environmental resources more effectively than native inhabitants. This efficient exploitation can lead to higher reproductive success and greater growth rates, which then allow nonnative species to biologically “swamp” resident fishes. The end result is replacement of native forms by nonindigenous species.

Many examples of these phenomena exist in the Southwest. Native Gila topminnow (*Poeciliopsis occidentalis occidentalis*) have been eliminated through

aggression and predation by mosquito fish (*Gambusia affinis*) (Minckley and others 1977, Meffe and others 1983, Weedman and others 1996, 1997a,b). Brown trout (*Salmo trutta*), a native trout of Europe, have wider environmental tolerances, especially water temperature, and are more aggressive and piscivorous. They have replaced native trout across North America, and in the Southwest have been implicated in the demise of Rio Grande cutthroat trout (*Oncorhynchus clarki virginalis*), Gila trout (*O. gilae*), and Apache trout (*O. apache*) (Behnke and Zarn 1976). Red shiner has been implicated in the replacement of native fishes through competitive exclusion throughout low elevation grassland streams in the Southwest. Predation by species nonnative channel (*Ictalurus punctatus*) and flathead catfish (*Pylodictus olivaris*) directly prey on larval young and adult native fishes. Both of these species have been implicated in the demise of the razorback sucker (*Xyrauchen texanus*) (Marsh and Langhorst 1988). Some species of introduced chichlid have year-round breeding seasons. These fish reproductively swamp native fishes by sheer number, which appears also to be the case in the replacement of Rio Grande sucker (*Catostomus plebeius*) by nonnative white sucker (*Catostomus commersoni*) in the Rio Grande drainage of Colorado and New Mexico.

Habitat alteration, it is hypothesized, often allows nonnative fishes to dominate indigenous fish faunas (Maddux and others 1993). Minckley and Carufel (1967) concluded that red shiner contributed to the demise of native fish populations throughout the Southwest. In many areas Red shiners have replaced woundfin (*Plagopterus argentissimus*), spikedace (*Meda fulgida*), and loach minnow (*Rhinichthys (tiaroga) cobitis*), all of which are Federally listed species in Arizona and New Mexico (Deacon 1988, Marsh and others 1989, Douglas and others 1994). While the mechanisms of replacement are not known, Rinne (1991) and Douglas and others (1994) indicated that red shiner adults occupied habitats used by juvenile spikedace and loach minnow in the Verde River. Nonnative trout have been shown to have detrimental effects on the Little Colorado River spinedace (*Lepidomeda vittata vittata*) in terms of predation and displacement (Blinn and others 1993). Centrarchids, especially, have been problematic for native fishes. This taxon has been introduced throughout Southwestern streams for sport, and by nature they are predacious, which has had severe impacts on all native forms. In the Verde River of Arizona, smallmouth bass (*Micropterus dolomieu*) and Green sunfish (*Lepomis cyanellus*) have eliminated a complete native fish fauna (Rinne and others 1999).

Similar to continental trends, establishment of nonnative fishes and replacement of native fishes has occurred on a regional scale throughout the Southwestern grasslands since the late 1800s (Rinne

1990, Fuller and others 1999). Nonnative fishes were imported for sport, biological control, and bait along with accidental introductions of unwanted species (Deacon and others 1964). Currently, 84 nonindigenous fishes are now found in the waters of Arizona and 78 in New Mexico (tables 5-4 and 5-5) (Minckley 1973, Hanson 1980, Sublette and others 1990, Fuller and others 1999). In New Mexico, introduced fishes account for more than half of the total fish fauna in the Rio Grande, Mimbres, Tularosa, San Juan, Zuni, San Francisco, and Gila Rivers (Propst and others 1987, Platania 1991, Sublette and others 1990, Rinne and Platania 1995, Propst 1999). In the Pecos and Canadian Rivers, nonnative fishes comprise 42 percent and 48 percent of the current fish fauna, respectively (Sublette and others 1990). Similar effects are seen in Arizona. Nonnative fishes comprise 82 percent of the fauna in the Colorado River basin, 68 percent in the Little Colorado, 50 percent in the San Juan, 40 percent in the Gila, 38 percent in the Bill Williams, 25 percent in the Rio Yaqui basin, and 10 percent in the Virgin River. The Sonotya and Rio Magdalena basins are free of nonnative fishes.

Native fish of the montane and grassland reaches of the Eastern Rocky Mountains and Western North America may be at greater risk from the introduction of nonnative species than native fishes in other parts of the country because of their isolated evolutionary history (Moyle and others 1986, Minckley and Douglas 1991). Additionally, depauperate native Western species live in dynamic watershed and aquatic environments, both of which make them vulnerable to replacement by nonnative fishes (Minckley and Deacon 1968, Moyle and Nichols 1973, 1974, Moyle 1976). Many native fishes in the West have not developed protective mechanisms against predation, general competition, and genetic hybridization, and they have low tolerance to biological interactions with nonnative fishes (Miller 1961, Deacon and Minckley 1974, Deacon 1979, Minckley and others 1986, Smith and Miller 1986, Sublette and others 1990). To a large degree, many Southwestern fishes exist as relict populations in single habitats, which exacerbates their precarious state (Deacon and Minckley 1974).

Effects of nonnative fishes on native fish communities are not well understood and are typically observed from major alterations in the native fish fauna. Direct effects from the interactions with nonnative fishes come in the form of predation (Schoenherr 1981, Minckley 1983, Blinn and others 1993), competition (Deacon and others 1964), hybridization (Rinne and Minckley 1985, Dowling and Childs 1992), elimination, reduced growth and survival, changes in community structure, and no effect (Moyle and others 1986). Impacts on various life stages also occur in time and space. Indirect effects, such as stress or other behavioral interactions are more difficult to document. The outcome of many of these

Table 5-4. Current distribution introduced nonnative grassland fishes by drainage basin in New Mexico. (Table adapted from Sublette and others 1990)

Species	Rio Grande	Canadian	Pecos	San Juan	Mimbres	Tularosa	Gila
Threadfin shad	X	X	X	X			
Cutthroat trout	X	X	X	X			
Coho Salmon	X	X	X	X			
Rainbow trout	X	X	X	X	X		X
Kokanee salmon	X	X	X	X			X
Brown trout	X	X	X	X	X		X
Brook trout	X	X	X		X		X
Lake trout	X						
Northern pike	X	X	X	X			
Zebra danio	X						
Goldfish			X				
Grass carp							
Carp	X	X	X		X		X
Fathead minnow	X	X	X	X	X		X
Bullhead minnow	X						
Yellow bullhead	X	X					X
Guppy	X						
Sailfin molly	X						
Inland silverside			X				
White bass	X	X	X				
Striped bass	X		X				
Rock bass			X				
Warmouth	X		X				
Smallmouth bass	X	X	X	X			X
Spotted bass			X				
White crappie	X	X	X		X		X
Black crappie	X	X	X				
Walleye	X	X	X				

Key: X = presence.

Table 5-5. Current distribution of nonnative grassland fishes by drainage basin in Arizona.

Species	Colorado	Little Colorado	Bill Williams	San Juan	Sonoyta	Yaqui	Gila	Virgin
White sturgeon								
American eel	X		X					
Threadfin shad	X						X	
Coho salmon								
Sockeye salmon								
Rainbow trout	X	X		X			X	
Cutthroat trout								
Brown trout								
Brook trout								
Arctic grayling								
Northern pike								
Banded tetra	X ?							
Carp	X	X	X				X	X
Goldfish								
Grass carp								
Golden shiner	X	X		X			X	

Table 5-5. Continued.

Species	Colorado	Little Colorado	Bill Williams	San Juan	Sonoyta	Yaqui	Gila	Virgin
Utah chub	X ?							
Red shiner	X	X	X				X	X
Redside shiner	X ?						X	X ?
Sand shiner		X ?						
Fathead minnow	X	X	X			X	X	
Bigmouth buffalo							X	
Black buffalo							X	
Smallmouth buffalo							X	
Flathead catfish	X						X	
Channel catfish	X	X	X				X	
Blue catfish	X ?							
Black bullhead	X	X				X	X	
Yellow bullhead	X	X					X	
Brown bullhead	X							
Plains Killifish	X	X						
Mosquitofish	X	X	X			X	X	X
Pacu								
Variable platyfish								
Green swordtail								
Sailfin molly	X						X	
Mexican molly	X						X	
Guppy	X						X	
Striped bass	X							
White bass	X							
Yellow bass							X	
Smallmouth bass	X						X	
Spotted bass	X						X	
Largemouth bass	X							
Warmouth								
Green sunfish	X	X	X				X	X
Bluegill	X	X	X			X	X	
Redear sunfish	X						X	
Pumpkinseed							X	
Rockbass								
White crappie							X	
Black crappie	X	X					X	
Sacramento perch								
Walleye							X	
Yellow perch	X	X					X	
Sargo								
Bairdiella								
Corvina								
Convict cichlid							X	
Mozambique mouthbrooder	X						X	
Blue tilapia								
Red-breasted tilapia							X	
Striped mullet	X							
Spotted sleeper	X							
Longjaw mudsleeper	X							
Mottled sculpin	X							

Key: X = presence. X? = questionable.

negative interactions is that remaining populations of native fish are often isolated, decline to low levels leading to categorization as threatened, endangered, or at risk (Miller 1961, Deacon and others 1979, Williams and others 1985, Miller and others 1989, Rinne and Minckley 1991).

Habitat Threats

General Landscape Change and Habitat Loss

Beginning in the 1880s, marked ecological changes over the Southwestern landscape occurred. By the 1890s, regional degradation was evident (Hasting and Tuner 1965). Dense stands of Southwestern grasslands, lush riparian areas, free flowing rivers and unaltered watersheds that were in place at the time of the Spanish explorers and later by Anglo settlers, had long since been irrevocably altered by livestock grazing and the damming and diverting of river water for human consumption (Griffiths 1904, Mehrhoff 1955, Scurlock 1998, Ambos and others 2002). Cottonwood gallery forests that once lined Southwestern rivers died due to channel incision and dropping water tables and were being replaced by nonnative phreatophytic species such as salt cedar (*Tamarisk* sp.).

Examples were noted throughout the Southwest. The San Pedro River, once a slow moving river with cienegas, had been changed to a steep-walled entrenched channel through which water moved rapidly, thus transporting large amounts of sediment into the Gila River. By the end of the 19th century, the once perennial San Pedro River had changed to an intermittent system exhibiting dry reaches throughout the year. Events such as these were relatively unknown prior to 1890 (Hastings and Turner 1965). In the grasslands of New Mexico, similar events occurred in the Rio Puerco (Rio Grande drainage) and Rio Chaco (San Juan drainage) watersheds (Bryan 1928, Scurlock 1998).

Water Use

The original aquatic habitats of the Southwestern grasslands once consisted of large rivers (5th order), streams, cienegas, and springs. Since settlement by Native Americans and Euro-Americans, these resources have undergone extreme modification (Leopold and others 1964, Hastings and Turner 1965, Sublette and others 1990, Minckley and Deacon 1991). Water development and its use for agriculture began as early as 900 A.D. when pueblo dwelling Native Americans collected surface runoff or diverted water from Southwestern rivers for agriculture and domestic uses. As the Spanish-Americans began settling the region (approximately 1521), increased demand was

placed on limited Southwestern waters resources (Dozier 1983, Carlson 1990, Scurlock 1998). The use of metal tools, domesticated animals, and engineering skills used by Spanish-American settlers made even a greater impact on water resources in the region. In New Mexico, the *acequia* or irrigation ditch was so important that they were dug prior to the construction of human dwellings. By the end of the 1600s, all or portions of present-day New Mexico's perennial streams were being diverted into irrigation networks (Scurlock 1998). As Anglo-Americans (nonnative, non-Hispanic peoples) entered the Southwest in the early 1820s, more demand was laced on water resources for expanding commercial agricultural and economic interest (Hastings 1959, Scurlock 1998). New irrigation systems were constructed, and older systems expanded by private companies whose economic well being was tied directly to the growing economy (Briggs and Van Ness 1987, Clark 1987, Scurlock 1998). Technological advances allowed wells to be dug that tapped underlying aquifers. Windmills also gave access to underground water resources. By 1880 water shortages had begun in the Southwest (Westphall 1965).

Large-scale change in natural stream flow and surface water dispersion occurred during the latter part of the 19th and beginning of the 20th centuries when dams were built on the major rivers of New Mexico and Arizona (Mueller and Marsh 2002). Then, with the passage of the Reclamation Act of 1903, the free-flowing waters and cienegas of the American Southwest were changed forever (Hastings and Tuner 1965). The perennial shortage of water for human use inherent in the region along with the post-World War II population boom precipitated the need for larger dams and diversion systems. Large-scale dams and diversion projects were built to store water, control sedimentation and flooding, generate power, and disperse water for irrigation. Today, dams and diversions are found on all major rivers and many of their tributaries throughout the Southwest (fig. 5-3 and 5-4). Large, artificial impoundments now make up the major surface waters in the Southwestern grasslands (Minckley 1973, Sublette and others 1990). Expanding agriculture, especially in the lower elevation grassland valleys, has necessitated the pumping of groundwater because of the shortages in surface water allocations; often times the result is the extraction of water that exceeds rates at which water recharges. Some aquifers are essentially being "mined" for their fossil water. Pumping of aquifers has placed many rivers and springs in the grasslands of the Western United States at risk by lowering regional and local water tables.

Interbasin transfer projects, such as the Central Arizona Project (fig. 5-5) now transport water across hundreds of kilometers into foreign basins. Natural drought cycles (one of which at publication time is in

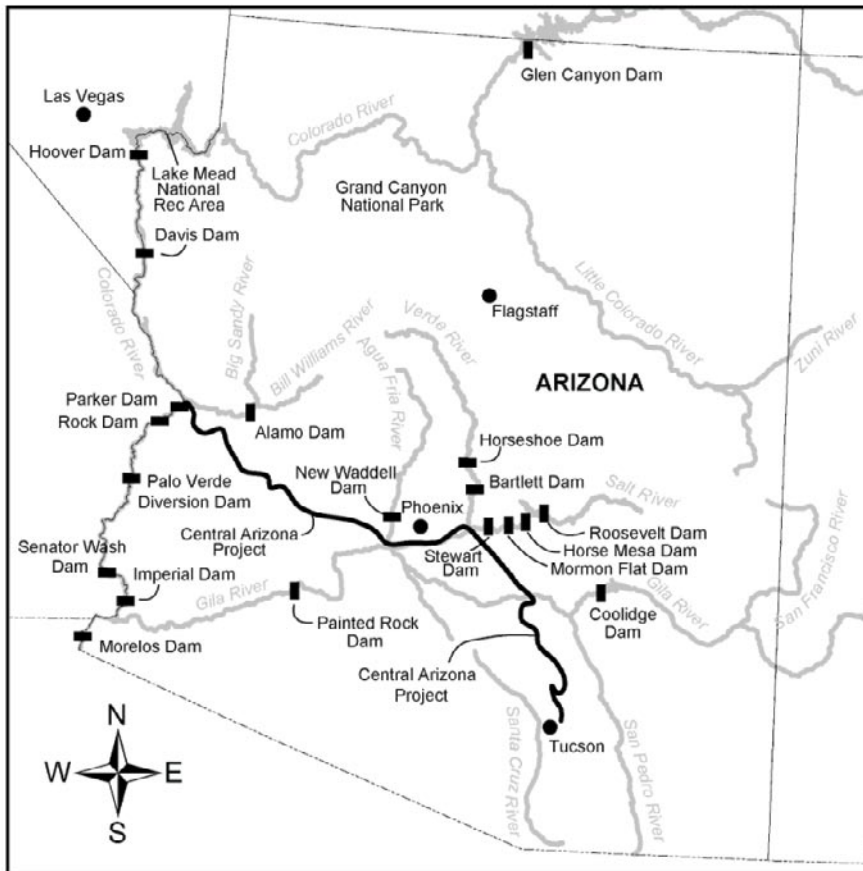


Figure 5-3. Major dams and diversions on rivers in Arizona.



Figure 5-4. Major dams and diversions on rivers in New Mexico.

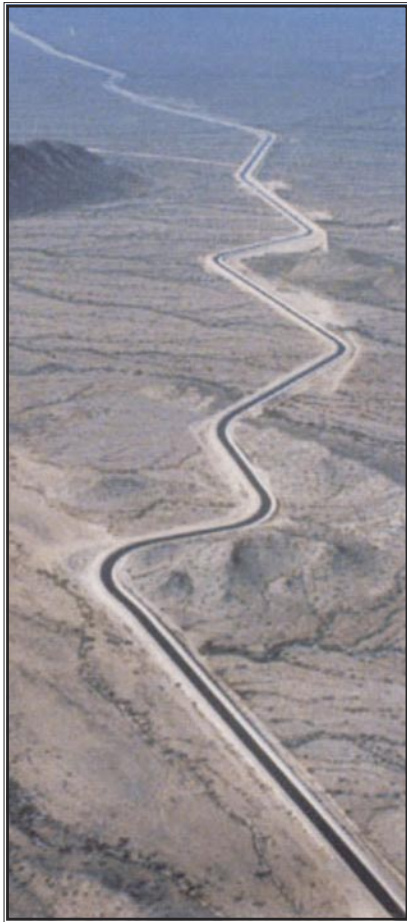


Figure 5-5. Central Arizona Project.

progress in the Southwest) add additional stress to a system that is largely overallocated and overused. Though completed with the best of intentions, all of these activities have disrupted the natural flows of grassland rivers and altered their physical properties. Today much of Arizona's and New Mexico's grassland river flows are found in canals, ditches, and drains.

In New Mexico, major dams or diversions exist on every large mainstem river in the State (table 5-6; fig. 5-4). There are also 32 small watershed projects with 71 flood control dams and 41 miles of diversions that were constructed since 1957. On the Rio Grande there are 13 major dams and diversions in New Mexico. Along the upper and middle reaches of the Rio Grande, six dams and three major diversions have been built. There are 321 km of canals (9.6 km concrete lined), 928 km of laterals (6 km lined), and 648 km of open and concrete pipe drains. The objective of these projects is to provide irrigation water, control sediment, and prevent flooding in the Middle Rio Grande Valley. Over 36,000 ha of agricultural lands are irrigated by water diverted from this reach of the Rio Grande. On the lower Rio Grande in New Mexico and West Texas, there are two major dams, six diversion dams, 222 km of canals,

Table 5-6. Dams and diversions on major rivers in the New Mexico.

Structure	River	Date built
Heron	Rio Grande	1971
El Vado	Rio Grande	1935
Abiquiu Reservoir	Rio Grande	1963
Cochiti Dam	Rio Grande	1975
Galisteo Dam	Rio Grande	1970
Nambe Falls Dam	Rio Nambe	1974
Jemez Canyon Dam	Jemez	1953
Angostura Diversion	Rio Grande	1938
Isleta Diversion	Rio Grande	1934
San Acacia Diversion	Rio Grande	1934
Elephant Butte Dam	Rio Grande	1916
Caballo Dam	Rio Grande	1938
Leasburg Diversion	Rio Grande	1919
McMillan Dam	Pecos	1908
Brantley Dam	Pecos	1991
Avalon Dam	Pecos	1888
Sumner Dam	Pecos	1937

731 km miles of laterals, 744 km of drains, and one hydroelectric power generating facility. Most drains in this reach of river are unlined earthen structures. These structures provide irrigation water for 79,320 ha of farm land in addition to power generation for cities and industry in the region. Water is also provided to irrigate 10,117 ha in the Juarez Valley, Mexico. Today, the lower Rio Grande is essentially a conveyance ditch from the headwaters of Elephant Butte to the Texas border. In the grassland reaches of the Pecos River, four dams have been constructed in New Mexico. There are 242 km of laterals, 59 km of canals, and about 38 km of drains in this system. Water stored in Pecos River reservoirs irrigates almost 10,000 ha in the region.

As with New Mexico, all major rivers and many tributaries in Arizona now are impounded, controlled, diverted, and in some cases overallocated (table 5-7; fig. 5-3). On the Colorado River there six dams, on the Bill Williams River one, four on the Salt River, two on the Verde River, and two on the Gila River. Impoundments on the Colorado River and its tributaries now account for the major surface waters in Arizona. The Colorado River—the once turbid and swift-flowing water that meandered through braided channel system bordered by riparian vegetation and marshes—has now been dammed, diverted, channelized, and confined by levees (Mueler and Marsh 2002). The largest artificial bodies of water in Arizona—Lakes Havasu, Powell, Mead, and Mohave—are all located on the Colorado River. Remote reaches of the Colorado River retain characteristics of a free-flowing stream as does the Salt River Canyon and the upper reaches of the Verde River.

Effect on Fishes—Water, above all, is of course the key habitat element needed to sustain fishes (Heede

Table 5-7. Major dams and diversions of the Lower Colorado River Basin, Arizona.

Structure	River	Date built
Granite Reef Diversion	Salt River	1908
Theodore Roosevelt Dam	Salt River	1911
Horse Mesa Dam	Salt River	1927
Mormon Flat Dam	Salt River	1925
Stewart Mountain Dam	Salt River	1930
Horseshoe Dam	Verde River	1946
Bartlett Dam	Verde River	1939
Laguna Dam	Colorado River	1909
Boulder Dam	Colorado River	1936
Imperial Dam	Colorado River	1938
Parker Dam	Colorado River	1938
Headgate Rock Diversion	Colorado River	1944
Palo Verde Diversion	Colorado River	1957
Coolidge Dam	Gila River	1928
Painted Rock Dam	Gila River	1960
New Waddell Dam	Agua Fria River	1928
Alamo Dam	Bill Williams River	1968

and Rinne 1990, Rinne 1990, 2000). The impact of impounding, diverting, dewatering, and draining of Southwestern waters along with mining of ancient aquifers has eliminated or drastically altered the aquatic habitats needed for survival of native Southwestern grassland fishes. The affect of these activities has accelerated the early decimation of native fishes found in grassland reaches throughout Southwestern rivers (Minckley and Deacon 1968).

Damming—Dams present a suite of problems for native fishes. First order impacts begin soon after the dam is in place (Neel 1963, Orth and White 1993). The physical structure acts as a barrier to upstream migration, and the impoundment can prevent downstream migration. Natural cycles of flooding and drought flows are now determined by power and water demands of cities and agriculture (Johnson and Rinne 1982). Other first order effects are altered sediment load, changes in water quality, and plankton (fig. 5-6). Second order impacts alter channel morphology and change substrate composition. Third order impacts relate to the cumulative effects of first and second order effects. It is the third order effects that impact fish and invertebrates. The physical structure interferes or prevents upstream migration, and often times the impoundment above precludes downstream migration due to vastly different water quality than preimpoundment conditions. For example, the Rio Grande once had large runs of female American eel (*Anguilla rostrata*). Other species affected by dams on the Rio Grande and Pecos Rivers were freshwater drum (*Aplodintus grunniens*), river carpsucker (*Carpoides carpio*), blue sucker (*Cycheptus elongates*), grey redhorse (*Moxostoma congestum*). Dams on the Lower Colorado system have

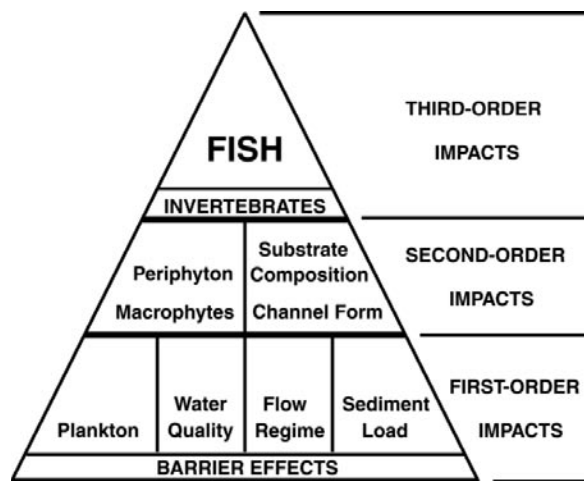


Figure 5-6. Impacts of impoundments on riverine systems.

been particularly destructive to the long-lived big river fishes such as flannelmouth sucker (*Catostomous latipinnis*), razorback sucker (*Xyrauchen texanus*), Colorado pikeminnow (*Ptychocheilus lucius*), humpback chub (*Gila cypha*), and bonytailed chub (*G. elegans*). Dams alter the riverine environment upstream and downstream of the structure. Stratification and point of release from the dam also determine water quality and temperature (fig. 5-7). For example, hypolimnetic releases have low relative water temperatures, are often anoxic, and can also have elevated nutrient, iron, manganese, and hydrogen sulfide concentrations (Reid and Wood 1976, Orth and White 1993). These conditions may be unsuitable for native fishes and native macroinvertebrates, but may be excellent for introduced cool-water game fishes. In addition to unfavorable habitat conditions, native fish now face the threat of predation and competition from these alien forms. Releases such as these occur at Stewart Mountain Dam on the Salt River and Caballo Dam on

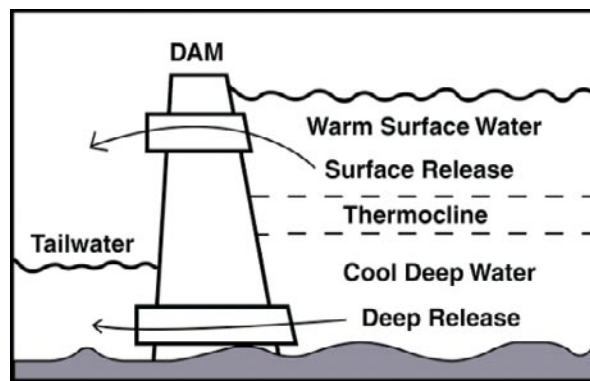


Figure 5-7. Impacts of dams on upstream and downstream water temperature.

the Rio Grande. Water chemistry and conditions that once favored native grassland fishes below Elephant Butte and Caballo Reservoirs now support species cool-water species such as trout and walleye (*Stizostedion vitreum*), along with various sunfishes and catfishes. In many reaches of Southwestern rivers (such as the Rio Grande, Salt, Colorado), dams have dewatered hundreds of kilometers of river channel while permanently inundating others. Flood plain areas that may have acted as nursery or rearing areas are now inaccessible along most major rivers.

Diversion—Diversion of stream waters during irrigation season desiccates many riverine habitats causing local extirpations and extinctions in reaches that have become permanently dewatered. Impacts are more severe during drought years. For example, Rio Grande silvery minnow living below the San Acia diversion on the Rio Grande often run the risk of coping with decreasing flows and drying of the channel during drought years. Fish are unable to ascend the diversion as larvae. As the stream dries many individuals are lost and recruitment is reduced. Similar situations occur on the Salt River below Stewart Mountain Dam and in the lower reaches of the Colorado River near the Mexican border. During irrigation season many fishes become stranded in irrigation canals as water flows are decreased and headgates are closed. The effects of dewatering appear too be more pronounced in the middle to lower elevations of Southwestern streams (fig. 5-3 and 5-4). These reaches flow through grassland and desert ecotypes that are farther from headwater sources and are more prone to natural dewatering. The placement of diversion systems above these reaches increases the probability of dewatering. Canals and laterals often times are earthen structures, but many miles of these are concrete lined. Lining canals with concrete, while increasing irrigation efficiency, prevents water from filtering into the water table. Seepage from unlined irrigation systems recharges aquifers, flows into drains, and recharges rivers from which it was diverted, thus supporting fish and riparian communities.

Earthen canals are regularly dredged to clean and maintain efficient flow of water. During this operation, riparian vegetation is either partially or completely removed so that equipment can complete the dredging operation. Fish communities in these systems (common along the Ro Grande) often harbor native fishes. Applications of herbicides to control aquatic weeds and pesticides to control mosquitoes are common. Such chemicals alter water quality and often make conditions uninhabitable for certain fishes (Minckley and Deacon 1968).

Irrigation systems present a suite of problems for native fishes that once were abundant in the unaltered river systems from which present systems are watered. The irrigation systems that are often lined with concrete

and are highly regulated cause organic components that would typically be found in natural riverine systems to be almost nonexistent in the artificial environment. Mechanical and chemical vegetation removal and dredging of such systems are regularly conducted to clean, facilitate flow, and effect efficient transfer of the precious commodity—water. The threat of dewatering and drying is ever present. Even in primitive, earthen irrigation canals, aquatic habitats have been highly simplified, flow regimes altered, and the fish community impacted through the introduction of nonnative fishes and other alien aquatic species such as crayfish and other invertebrates. Though these problems for native fish are salient, irrigation systems do allow fishes to persist, even if that occurs at reduced levels. The drains that parallel the Rio Grande in New Mexico for all practical purposes harbor the fish community of its middle and lower reaches.

Pumping—Pumping of groundwater lowers water tables; this causes interruption in arid land streams. Springs and marshes can also become desiccated from lowering of water tables. The upshot is the demise of many aquatic habitats and the loss of native fishes populations. One of the classic examples of this, while not in the grasslands reaches of Arizona or New Mexico, was the near extirpation of the Devil's Hole pupfish (*Cyprinodon diabolis*). In 1969, corporate farm interests intended to pump the Ash Meadows aquifer, located in southeastern Nevada near Las Vegas, to increase farm production. The result would have been the lowering of the water table, which in turn would dewater the spring, ultimately leading to the extinction of this indigenous endemic pupfish species. Fortunately, for the Devil's Hole pupfish, concerned scientists and citizens were able to bring a halt to this action.

Grazing, Riparian Corridors, and Fish Relationships

The ecological conditions of Southwestern watersheds and riparian corridors have undergone dramatic changes since the settlement of the region by Native Americans and Euro-Americans (Carton and others 2000). Many of these changes have been attributed to grazing of domestic livestock over the past 460 years. The first livestock grazed in the region were part of the Spanish expeditions that entered the region in 1539. Spanish settlers grazed 4,000 sheep, 1,000 cattle, 1,000 goats, and 150 mares with colts in the Middle Rio Grande reach of present-day New Mexico around 1598 (Baxter 1987, Bayer and Montoya 1994, Scurlock 1998, Floyd and others 2003). Overgrazing was observed in the area as early as 1630 (Stewart 1936, Scurlock 1998). From 1785 to 1900 some 6,000 goats grazed the foothills of the Sandia Mountains. These lands later came under the jurisdiction of the

Cibola National Forest (Scurlock 1998). In addition to Spanish settlers, many contemporary Native American peoples grazed livestock. As many as 30,000 head of sheep grazed the lands of the Hopi Pueblos during the 1700s (Schickendanz 1980). By 1820 there was an estimated one million sheep grazing in New Mexico, and by the mid 1800s about three million sheep grazed the grasslands and riparian areas of that region (Baxter 1987, Scurlock 1986, 1998). The ending of hostilities between the United States government and Native Americans along with the completion of transcontinental railways and their spur routes led to sharp increases in livestock from 1870 to 1890 (Scurlock 1998). By 1880 five million sheep and 250,000 head of cattle grazed the grasslands and river valleys of New Mexico (Bailey and Bailey 1986).

Arizona experienced a long history of grazing by domestic livestock similar to New Mexico (Young 1998, Wildeman and Brock 2000). Spanish horses were first grazed in present-day Arizona in 1515, cattle in 1521. Goats and sheep, imported by Spanish settlers and expeditions, were grazed on the Arizona grasslands by 1700 (Barnes 1926, Peplow 1958). Peplow (1958) reported that the most important (and probably the most well stocked) ranches were at present-day San Bernardino, Babocomari, San Pedro, Arivaca, Calabasas, Saporí, Radenton, San Rafael de la Zanje, Sonoita, Tubac, and ranches in the San Simon Valley, Aqua Prieta, and Pueblo Viejo. After wars with the Apache and Navajo Tribes, most if not all of the haciendas were abandoned (Wildeman and Brock 2000). Subsequently, large herds of domestic livestock roamed freely across the grasslands of Arizona.

Anglo settlement and livestock grazing began in the early 1870s (Young 1998). Cattle driven from Texas were grazed the Salt River watershed and 100,000 head of cattle grazed the headwaters of the San Pedro River in southern Arizona (Wildeman and Brock 2000). By 1891 about 1.5 million cattle grazed the grasslands and watersheds of southern Arizona (Cox and others 1983, Ferguson and Ferguson 1983, Hendrickson and Minckley 1985).

By the early 1870s, even untrained observers noted that range conditions across the Southwest were rapidly deteriorating. Overstocking of upland and riparian grassland habitats and drought (late 1880s) was implicated to be the primary cause of decline (Leopold 1946, Hastings 1959, Gresswell and others 1989, Scurlock 1998, Kruse and Jemison 2000). Arroyo cutting and erosion of topsoil began to occur as vegetation was stripped from watershed slopes. Overutilized riparian areas, which had formerly acted as filter strips in their pristine state, were unable to trap the large quantities of soil moving down slope into stream channels (Hastings 1959, Miller 1961, York and Dick-Peddie 1969, Dick-Peddie 1993, Minckley 1969, Sublette and

others 1990, Bahre 1991, 1995, Friedman and others 1997, Wildeman and Brock 2000). Southwestern grasslands east of the Rocky Mountains (Great Plains type grasslands) fared better than desert grasslands of western New Mexico and Arizona. Great Plains type grasslands evolved under the grazing pressure of large herds of native ungulates (Milchunas and others 1988, Loftin and others 2000).

In contrast, desert grasslands west of the Continental Divide evolved under light grazing pressure from native ungulates. Thus, making the latter susceptible to degradation by livestock grazing (Elmore 1992, Bock and Bock 1993, Loftin and others 2000). Grassland reaches of streams at intermediate and higher elevations in the American Southwest suffered far less impact than middle and lower elevation grassland stream reaches. Many of these stream reaches are found within National Forest System lands and Native American lands. Greater amounts of precipitation, relative to lower elevation grasslands, made these riparian zones more resistant to the effects of grazing and other land use practices. Steeper gradient reaches also are less accessible to grazing livestock and use by agriculture.

As one can see, the long history of grazing in the Southwest has had definite impacts on Southwestern watersheds. Today, many riparian areas in the Southwest are considered to be functionally impaired due to environmental stressors such as overgrazing (Leopold 1974, Gallizioli 1977, Davis 1977, Hibbert 1979, Platts 1983, Clarkson and Wilson 1995, Patten 1998, Baker and others 2000). Grazing of livestock affects watershed condition and function through the direct removal of vegetation and the mechanical action of trampling (Johnston 1962, Johnson and others 1978, Tromble and others 1974, Hibbert 1979, Ryder 1980, Reid 1993, Peiper 1994, Belsky and others 1999). Removal of riparian vegetation and trampling through grazing can reduce bank stability and the channel-narrowing capacity of a stream (Platts 1979, 1982, Platts and others 1983, Abdel-Magid and others 1987a,b, Medina 1995, Neary and Medina 1995, Clary and Kinney 2000). Hoof action disturbs stream substrate, loosens and/or compacts soils, which in turn reduces infiltration, increases surface runoff, and accelerates movement of soils into watersheds (Wilcox and Wood 1988, Neary and Medina 1996, Payne and Lapointe 1997, Loftin and others 2000). Typical changes in stream morphology and aquatic habitat are streambank sloughing, widening and shallowing of the streambed, channel entrenchment, velocity reduction (Behnke and Raleigh 1978, Platts 1981a,b, Allen-Diaz and others 1998). The result is a less complex aquatic habitat. Fine sediments (less than 2 mm) that would typically be suspended in the water column and transported downstream are deposited on

the streambed as water velocity decreases. Erosion of bank materials, due to loss of ground cover and increased surface runoff, adds to the fine sediment portion of the stream bedload (Dune and Leopold 1978, Pogacnik and Marlow 1983, Williams and Sada 1985, Abdel-Magid and others 1987b, Schultz and Leininger 1990, Platts 1991, Medina 1995, Neary and Medina 1996). Fines in bedload fill interstitial spaces and sedimentation of channel substrate begins (Lisle 1989, Bevenger and King 1995). Increases in temperature can occur due to alteration of species composition and reduction in plant density through direct removal of vegetation by grazing livestock (Amaranthus and others 1989, Schultz and Leininger 1990). Overgrazing of livestock of watersheds in arid and semiarid regions such as the Southwestern grasslands may have more severe consequences to watersheds, riparian habitats, and their inhabitants than in more mesic habitats where plant vigor is assumed to be greater (Behnke and Raleigh 1978).

Physical and chemical changes caused by grazing in grassland watersheds, riparian areas, and instream habitats clearly may have a direct affect on the quality and quantity of fish habitat (Meehan and Platts 1978, Platts and Nelson 1985, Platts 1981a,b, 1991, Bestchta and Platts 1986). Although the linkage between fish and grazing is not well defined and many cases cannot be correlated (Rinne 1988a,b, 1999a,b, 2000), evidence does suggest that as grazing negatively impacts fish habitat, it also affects fish populations (Stuber 1985). For example, Platts (1981a) reported that a stream reach receiving high intensity grazing from sheep had a channel width five times greater, stream depth one-fifth as deep, and stream shore depth one-third as deep in heavily grazed reaches compared to moderately or lightly grazed adjoining reaches. Livestock grazing in riparian areas can reduce overhanging banks that can serve as cover for fish, and cause stream banks to become reposed with a decrease in stream shore depth (Overton and others 1994). Platts (1991) concluded, as did Behnke (1977), that degradation of riparian habitats through grazing was one of the major vectors causing the decline of native fishes in the Western United States. Grazing causes temporal and spatial changes in the riparian canopy and composition, which in turn may elevate stream temperature, and thus may have negative effects on the persistence or productivity of some fish populations, particularly cold water species (Meehan and others 1977, Meehan 1996). For example, Platts and Nelson (1989a,b) found that salmonid biomass was correlated with riparian canopy density in Great Basin streams. The authors suggested that solar insolation was a limiting factor in these waters. Sponholtz and Rinne (1997) reported that on reaches that were grazed by livestock in the Verde River, Arizona streambank vegetation was 60

percent lower, channel width 50 percent greater, and stream velocity was 85 percent lower than in grazed reaches. Fish density in the grazed reaches was 50 percent lower than at the ungrazed sites.

Although fine sediment (less than 2 mm) is a natural component of bedload (Hynes 1972), an overabundance of fines can affect the quality and complexity of aquatic environments and its inhabitants (Downes and others 1998). Abiotic changes (that is, sedimentation) can decrease the diversity and abundance of macroinvertebrates that serve as food for fishes (Brown and Moyle 1991). Allen-Diaz and Jackson (1998) reported that insect family richness was significantly reduced on moderately grazed plot versus lightly or ungrazed plots. Rinne (1988a) reported similar findings in the Rio De Las Vacas, Santa Fe National Forest, New Mexico. Fine sediment can also impair reproductive success of gravel spawning fish. Meehan (1991) indicated that excessive percentage of fines negatively impacted the reproduction and recruitment in salmonid populations. In contrast, Rinne (1999b) could not identify if fines had negative impacts on spikedace and loach minnow and could not say it was a major limiting factor. Abundant populations of these two species are found in the Gila-Cliff Valley, New Mexico, where grazing is present (Propst and others 1986). Further, Rinne (2000) cautions that often times we are dealing with difficult-to-determine thresholds and discrete percentages of fines that negatively affect various stages of a fish life cycle. Too, streams with a greater percentage of suitable spawning gravels may be able to tolerate higher levels of sedimentation and still recruit fish into the population.

There is a scarcity of research on grazing effects on Southwestern fishes, especially nonsalmonids (Rinne 2000), and in some cases, when dealing with cypriniform species, grazing may not have an impact on this group of fishes. Although empirical information is scarce on the effects of grazing in arid land watersheds (Rinne 1999a, 2000), much research has been conducted in the Intermountain, Northern Rocky Mountain, and Pacific Northwest (Platts 1991, Rinne 1999a, 2000). This information can be used to lend insight into grazing effects in the Southwest. Platts (1991) and Rinne (1998, 1999a, 2000) both state that more scientific rigor and better study design were needed in past studies and will be needed in future research. Nonetheless, negative effects of grazing on watershed condition, riparian vegetation, and aquatic habitats alluded to above have been documented by many practitioners throughout the Western United States (Ames 1977, Platts 1979, 1982, 1991, Szarro 1989, Chaney and others 1990, Naimen and Descamps 1990, Elmore 1992, Belsky and others 1999). In a review article published in the *Journal of Soil and Water Conservation*, Belsky and others (1999) concluded that most scientific studies document

that livestock grazing continues to be detrimental to stream and riparian ecosystems in the West. It may well be that the linkages between grazing and its direct effect on native fishes are still obscure (Medina and Rinne 1999, Rinne 1999a,b, 2000). Nevertheless, both watershed health and the protection of native fishes should be our primary objectives. Managers and land owners should assess whether these two ecosystem components will be negatively affected before deciding to graze livestock in Southwestern watersheds.

Management and Conservation_____

The first step in management of fish is the recognition that a species is imperiled locally or nationally. Managers must have up-to-date lists of species so that appropriate action can be taken to implement conservation measures (Miller 1972, Pister 1976, 1981). Since the late 1960s States across the country along with the Federal government has been enacting legislation and enforcing protection of endangered species, subspecies, and critical habitat. Laws such as the Endangered Species Preservation Act of 1966, Endangered Species Conservation Act of 1969, and the Endangered Species Act of 1973 (amended in 1978 and 1979) were put in place with the purpose of conserving endangered and threatened species and their ecosystems (Johnson and Rinne 1982) as well as maintaining natural habitats and creating refugia for some critically threatened fishes (Miller 1972, Miller and Pister 1991). An example of this is the refugia created for the Owens Valley pupfish. In 1971, Congress established the Desert Pupfish National Monument to save threatened pupfishes living east of Death Valley (Deacon and Bunnell 1970). Similar actions could be put into play for fishes inhabiting Southwestern grasslands. Optimal habitat is infinitely more effective than imposition of restrictive regulations in the enhancement of fish populations especially with officially designated Threatened and Endangered species (Deacon and others 1979). Critical habitat has been designated for spikedace and loach minnow in the Southwest, and restocking programs for native fish are ongoing. Gila topminnow is being repatriated throughout its range in New Mexico and Arizona. Rio Grande sucker and chub are being planted into native waters in the upper Rio Grande basin, as are Rio Grande silvery minnow, razorback sucker, and Colorado pike minnow. These restockings have had mixed results due to the availability of optimum habitats or the continued competition with nonnative fishes. Nevertheless, these must be continued if we are to have any chance at successfully restoring native fishes.

Problems are inherent when trying to save native fish. First, because many native Southwestern fishes are nongame species considered by many, even in

resource agencies, as “rough fish,” garnering support from private land owners or individuals and companies leasing public lands can be a challenge. Second, funding conservation efforts for native fish is a challenge because of budgetary constraints, political climate of the time, and, again, the view of these fishes as somewhat less desirable than game fish. Third, the frequent lack of information and communication among scientists might even be encouraged by what is perceived as small “cliques” of researchers whose intention is development of their own view on how the species should be managed. Fourth, water is an ever-decreasing resource in the Southwest, and many water laws permit beneficial water use without regard to fish and wildlife.

Finally, what is especially grim are the logistics and realities of trying to preserve native fishes in Southwestern grasslands, especially fishes that inhabit the grassland reaches. The grassland reaches of most Southwestern streams occur at low elevations in broad river valleys that have been occupied by humans for a long time and that typically have the highest human concentrations. Major cities such as Albuquerque, Las Cruces, El Paso, Tucson, and Phoenix have significant impacts on the water supply and quality of these low elevation grassland reaches. Major water withdrawals occur in these reaches; development and alteration of stream morphology is extensive along with the removal of riparian habitats. Pollution—both chemical and biological (encroachment of nonnative aquatic species)—is an ever-increasing concern. There are probably hundreds of thousands of nonnative fishes in these large-order, low-elevation grassland reaches. The chance of removing nonnative fishes to ensure the survival of native forms is small at best and, being realistic, probably impossible. The best land managers, fishery professionals, and society at large could expect is to identify isolated reaches on smaller tributary streams where logistics and isolation might result in some chance of success. Recovery is viable in small, remote locations, but at a large scale loss will continue and native fish community diversity will decline (Johnson and Rinne 1982).

This is a sad note, but one must face realities when prioritizing conservation projects when funding and worker availability are often limited. With that in mind, the rest of this chapter looks at a number of native fish in detail.

Loach Minnow (*Rhinichthys (tiaroga) cobitis*)

Distribution and Population Trends—*Rhinichthys (tiaroga) cobitis* is endemic to the upper Gila River drainage of southwestern New Mexico, southeastern and east-central Arizona, and northeastern Sonora (Miller and Winn 1951, Koster 1957, Minckley 1973, Propst 1999). Its historic distribution in New Mexico

has been reported as the mainstem of the San Francisco River and Gila River along with many of the lower elevation tributaries (Koster 1957, Propst and others 1988). Minckley (1973) indicated that loach minnow were historically found in the Gila River above Aqua Fria Creek, Salt River, Verde River, San Pedro River, San Francisco River, and Blue River. In Mexico, the species was known from the upper reaches of the San Pedro River in northern Sonora (Miller and Winn 1951, Propst 1999). Today the range of *T. cobitis* has been much reduced and the population trend is declining. In Arizona, loach minnow remain in Aravaipa Creek, Eagle Creek, White River, Blue River, and the North Fork of East Black River; however, populations continue to decline over its range. Loach minnow was listed by New Mexico as a threatened species (19 NMAC 33.1) in 1975 and is Federally listed as a threatened species (USFWS 1986). In Arizona it is listed as a species of special concern (AZGFD 1996).

Preferred Habitat—*Rhinnichthys (tiaroga) cobitis* is found in moderate to large streams where gradients range from 1 to 3 or 4 percent (Rinne 1989, Rinne and Deason 2000). Loach minnow are a cryptic species associated with riffle habitats (Minckley 1973, Propst and others 1988, Propst and Bestgen 1991, Propst 1999) where they live in the interstitial spaces of large gravel and cobble substrate (LaBounty and Minckley 1972, Minckley 1973, Rinne 1989, Turner and Tafanelli 1983, Rinne and Deason 2000). Loach minnow have a reduced air bladder, streamlined body form, and large fins, which allow them to exploit high velocity, riffle habitats where it breeds and feeds on aquatic insects, principally fly and mayfly larvae and nymphs; however, stone, caddis and blackfly larvae are also part of loach minnow diet. The species moves in bursts of swimming from one substrate material to another much as do darters of the Eastern United States. Loach minnow are most closely associated with other native riffle dwelling species. It coinhabits similar aquatic space with desert suckers and speckled dace (Rinne 1992, Propst 1999).

Breeding Biology—In New Mexico, loach minnows reproduced during a 4 to 6 week period during late March and early June when waters temperatures reach 16 to 20 °C (Britt 1982, Propst and Bestgen 1991, Propst 1999). In contrast to New Mexico, Vives and Minckley (1990) found loach minnow spawning in the autumn in Aravipa Creek, Arizona. Bestgen and Propst (1991) indicate that runoff volume, timing, and duration of flow influences the onset of spawning for loach minnow. Loach minnow lay adhesive eggs on the undersides of rocks in flowing water habitats (42.7 cm/sec or less) and at depths of 6.1 to 21.3 cm (Britt 1982, Propst and Bestgen 1991, Propst and others 1988, Propst 1999). Propst and Bestgen (1991) observed that when water velocities slow (less than 5 cm/s), egg

mortality increased. Clutch size ranges from nine to 260 eggs per nest (Britt 1982, Propst and others 1988). Britt (1982) reported fecundity of females ranging from 144 to 1,200 eggs. Eggs of loach minnow hatch in 4 to 5 days at 21 °C (Propst and others 1988). Males may provide care during the incubation period (Propst and others 1985). Larvae average 5.4 mm TL at hatching (Propst 1999). Larvae use low velocity habitats after hatching (Propst and others 1988). Maximum life span of loach minnow ranges from 2 to 4 years (Minckley 1973, Britt 1982, Propst and others 1988).

Management Issues and Recommendations—

Two major factors appear to determine the persistence of loach minnow: presence of nonnative species and the degradation of historic loach minnow habitat. Propst (1999) reports that at least 15 nonnative fishes are in the New Mexico portion of the loach minnow range with a similar number occurring in its historic Arizona range. An eminent threat is from introduced catfishes (Family: *Ictaluridae*), principally flathead catfish (*Pylodictis olivaris*), channel catfish (*Ictalurus punctatus*), black bullhead (*I. melas*), yellow bullhead (*I. natalis*), and brown bullhead (*I. nebulosus*) (Propst and others 1988). Flathead catfish are highly piscivorous and become so at an early age. Channel catfish often feed in riffles, habitat used by loach minnow. Propst (1999) indicates that where channel and flathead catfish co-occur, loach minnow are rare or extirpated. In addition to these predacious species, nonnative centrarchids are established throughout the range of loach minnow (Minckley 1973, Sublette and others 1990).

Land use activities such as cattle grazing, timber harvest, and road building, all of which contribute fine sediments to river systems, have also been implicated in the decline of loach minnow. Fine sediments fill the interstitial spaces where loach minnow live, feed, and breed, thereby decreasing available habitat and impacting spawning success. Also, dewatering of streams for consumptive uses eliminates spawning areas and core habitat.

Management activities directed toward removal of nonnative fishes, decreasing fine sediment in watersheds, and maintaining free-flowing unaltered stream systems appear to be critical for the conservation of loach minnow, along with translocating populations into suitable habitats free of nonnative fishes (Propst 1999).

Rio Grande Silvery Minnow (*Hybognathus amarus*)

Distribution and Population Trends—Native to New Mexico, the species has been historically found in the Rio Grande drainage downstream from Velarde, NM, to the Gulf of Mexico, in the Rio Chama downstream from Abiquiu, and the Pecos River downstream



Rio Grande silvery minnow.

from Santa Rosa (absent in the reach from Red Bluff Reservoir to Sheffield) to its confluence with the Rio Grande in Texas. The Rio Grande silvery minnow also occurs in coastal drainages of Texas from the Brazos River west to the Rio Grande drainage of New Mexico (Sublette and others 1990, Bestgen and Platania 1991, Platania 1995, Propst 1999). Bestgen and Platania (1991) report *Hybognathus amarus* as formerly common in the Rio Grande from Cochiti to Socorro and in the Pecos River from Fort Summer to Carlsbad. Likewise it was common near the confluence of the Pecos and Rio Grande in Texas (Trevino-Robinson 1959, Edwards and Contreras-Balderas 1991). Rio Grande silvery minnow appear to have been extirpated in the Pecos River and the lower Rio Grande below Elephant Butte Reservoir, the former occurring in the 1970s and the latter by the 1950s (Propst and others 1987, Edwards and Contreras-Balderas 1991, Bestgen and Platania 1991, Propst 1999). Currently, *H. amarus* occupies less than 10 percent of its historic range (Propst 1999). The species is found only in perennial sections of the Rio Grande and incidentally in irrigation systems in the Middle Rio Grande reach from Cochiti to the head of Elephant Butte Reservoir (Sublette and others 1990, Platania 1991, 1993). Rio Grande Silvery minnow is listed by the USFWS as endangered (USFWS 2003), and by the New Mexico Department of Game and Fish as State Endangered, Group II (19 NMAC 33.1) (Propst 1999).

Preferred Habitat—Rio Grande silvery minnow are schooling fish and use a variety of habitats (Sublette and others 1990). Dudley and Platania (1997, 2000) report Rio Grande silvery minnow using mainstream habitats with water depths ranging from 0.2 to 0.8 m, water velocities of 0 to 30 cm/sec, and over silt/sand substrate. Propst (1999) indicates that a seasonal shift to slower velocity habitats with debris cover occurs in winter. While individuals are sometimes found in irrigation canals, often young and some adults have apparently been entrained during irrigation activities (Lang and Altenbach 1994, Propst 1999). Rio Grande silvery minnow are herbivores feeding on diatoms, larval insects skins, and live and decaying plant material (Sublette and others 1990).

Breeding Biology—Spawning activity of Rio Grande silvery minnow occurs at the end of spring and the beginning of summer with water temperatures ranging from 20 to 24 °C (Propst 1999). Platania and Altenbach (1998) report that the species is a pelagic spawner, and the eggs are semibuoyant and nonadhesive. Females are attended by multiple males with several encounters (Propst 1999). Females are reported as producing three to 18 clutches in a 12-hour period. Propst (1999) reports that the majority (more than 90 percent) of spawning individuals were age-1 adults. Further, it appears that mortality can be as high as 98 percent of breeding individuals (Propst 1999). Similar to other larval fishes, larvae of Rio Grande silvery minnow drift for several days and then migrate to low velocity habitats at the stream margins.

Management Issues and Recommendations—Propst (1999) lists factors related to the decline of Rio Grande silvery minnow. He suggests that nonnative fishes, impoundments, declining water quality, reduction of flow, and range fragmentation due to mainstem and diversion dams have together contributed to the decline and potential extirpation of Rio Grande silvery minnow in the Rio Grande drainage. Bestgen and Platania (1991) and Cook and others (1992) provide evidence that a combination of poor water quality, altered flow regimes, and hybridization with plains minnow, *Hybognathus placitus*, accounted for its elimination in the Pecos River.

Conservation strategies that maintain minimum flow, natural hydrographs, and improved water quality are needed to sustain Rio Grande silvery minnow. Maintenance of river flows will also alleviate loss through predation when minnows are concentrated in shrinking habitats.

Rio Grande Sucker (*Catostomus (Pantosteus) plebeius*)

Distribution and Population Trends—The Rio Grande sucker is native to Colorado, New Mexico, and Mexico. Historically, the species was widely distributed in the Rio Grande Basin of Colorado and New Mexico (Koster 1957, Sublette and others 1990, Langlois and others 1994). In New Mexico, its current distribution is the Rio Grande above the 36th parallel and its tributaries north of the 33rd parallel (Propst 1987, Sublette and others 1990, Platania 1991, Calamusso unpublished data). Rio Grande suckers are found in the Mimbres River, and introduced populations are established in the Rio Hondo (Pecos drainage), Gila River basin, and San Francisco drainage (Sublette and others 1990). Rio Grande sucker also inhabit six river basins draining four States of Mexico (Hendrickson and others 1980, Abarca and others 1995). The Rio Grande sucker appears to be declining across its range. At one

time the species was found in only one drainage in Colorado (Hot Creek) (Swift-Miller and others 1999a,b). In New Mexico, the species appears to be declining across its range (Michael Hatch, personal communication). Calamusso and others (2002) documented the decline and replacement of Rio Grande sucker by white sucker, *Catostomus commersoni*, in Rio Grande tributary streams in northern New Mexico. Rio Grande sucker was designated as State endangered by the Colorado legislature in 1993 when it was found that only one population remained (Swift and others 1999) in the State. While more common in New Mexico, Rio Grande sucker are considered rare in the Rio Grande mainstem (Rinne and Platania 1995) and is declining in the tributaries of the Middle and Upper Rio Grande basin (Calamusso 1992, 1996, Calamusso and Rinne 1996, 1999).

Breeding Biology—Temperature is the controlling factor as indicated by latitudinal and elevation differences in spawning time. Spawning begins in February in the southern portion of the species range and occurs progressively later northward (Smith 1966). Rio Grande sucker were found spawning in Animas Creek, Sierra County, NM, in February when water temperature reached 9 °C (Calamusso unpublished data) and in June in streams on the Santa Fe National Forest when water temperature also reached 9 °C (Calamusso 1996). Rinne (1995b) reported spawning in the Rio de las Vacas peaking in June during the declining spring flows, and Rausch (1963) found the species spawning in the Jemez River in May. In two streams on the Santa Fe National Forest, suckers spawned where dominant substrate ranged in size from 2.0 to 18.9 mm, water velocity ranged from 21.0 to 62.0 cm/s (mean 37.0 cm/s, and depth ranged from 9.0 to 28.2 cm (mean 16.2 cm) (Calamusso 1992). Koster (1957) indicated that Rio Grande sucker also spawned in the fall. Males and females become sexually mature at age 3 (Rausch 1963). Maximum life span of females is 7 years, with a corresponding mean standard length of 159 mm (Rausch 1963).

Preferred Habitat—Rio Grande sucker live in small to large middle elevation (2,000 to 2,600 m) streams over substrates that range from sand/small gravel to cobble (Sublette and others 1990). Calamusso (1996) and Calamusso and Rinne (in review) evaluated habitat use of Rio Grande sucker in six study streams on the Carson and Santa Fe National Forests of northern New Mexico. Rio Grande suckers were found in low gradient (3.2 percent or less) stream reaches and were associated with pool and glide habitat, low water column velocity (less than 20 cm/s), moderate depth (mean: 24.5 cm; range 7 to 62 cm), and cobble substrate. These habitats were used disproportionately to available habitats. Abundance of Rio Grande sucker was inversely related with stream gradient and mean water column velocity. Density and biomass of Rio

Grande sucker was positively related with increasing amounts of pool and glide habitat at the reach level, as was greater biomass.

Management Issues and Recommendations—

The Rio Grande sucker was the only catostomid endemic to the Rio Grande (Koster 1957, Sublette and others 1990, Langlois and others 1994). Rio Grande suckers probably once occupied all low gradient (less than 3.2 percent), middle-elevation tributaries to the Rio Grande in Colorado and New Mexico north of the 33rd parallel (Calamusso and others 2002). The decline in range and numbers of the Rio Grande sucker is related to introduction of nonnative fishes, especially white sucker, into the Rio Grande drainage (Calamusso 1996, Calamusso and Rinne 1999, Calamusso and others 2002). In all streams where Rio Grande sucker have been extirpated or are declining, white sucker are now present (Calamusso and others 2002). Although nonnative brown trout, *Salmo trutta*, are believed to prey upon Rio Grande suckers, Calamusso and Rinne (1999) could not detect a negative correlation between presence of the nonnative trout and the native sucker. Management directed at protecting remaining populations of Rio Grande sucker using fish migration barriers is warranted. In addition, Rio Grande sucker should be reintroduced along with Rio Grande cutthroat during restorations for the latter. Recovery efforts for both these species should consider the effect of nonnative species and stream gradient on their persistence and abundance when considering reintroduction sites.

Rio Grande Chub (*Gila pandora*)

Distribution and Population Trends—Rio Grande chub are native to the Rio Grande basin of Colorado, New Mexico, and Texas. In New Mexico the species is also found in the Pecos River and Canadian drainages, although it may have been introduced in the latter. While no recent systematic review of its status in New Mexico has been conducted, the species appears to be stable and widespread throughout the montane tributaries of the Middle and Upper Rio Grande basin. In contrast, Rio Grande chub is considered rare or may even be extirpated in the Rio Grande mainstem (Rinne and Platania 1995, Propst 1999) and in tributaries in the southern portion of the State below Albuquerque. Recent ichthyofaunal surveys of streams draining the east slopes of the Black range, Sierra County, NM, indicate that Rio Grande chub are being replaced by nonnative longfin dace, *Agosia chrysogaster*, (Calamusso unpublished data). Propst (1999) suggests that legal protection for the species in New Mexico may be warranted. In Colorado Rio Grande chub is considered to be in general decline and listed as threatened (Zuckerman 1983, Zuckerman and Langlois 1990, Sublette and others 1990).

Breeding Biology—Rio Grande chub spawn in spring and early summer (Sublette and others 1990). Rinne (1995a) found that spawning of Rio Grande chub reached its peak on the descending limb of the spring hydrograph in the Rio de las Vacas, New Mexico. Rio Grande chub are known to hybridize with longnose dace, *Rhinichthys cataractae*, when the two occur in reduced habitats due to dewatering or drought conditions (Cross and Minckley; 1960, Suttkus and Cashner 1981).

Preferred Habitat—Similar to many members of the Genus *Gila*, Rio Grande chub are found in low velocity habitats (pools, glides, runs) of small to moderate sized streams where gradients rarely exceed 2 percent and are often associated with instream woody debris or aquatic vegetation (Woodling 1985, Calamusso 1996). Rio Grande chub are midwater carnivores that, similar to many salmonids, feed on insect drift, zooplankton, and small fishes (Sublette and others 1990). *Gila pandora* also feed on detritus (Sublette and others 1990).

Management Issues and Recommendations—Declines in Rio Grande chub populations appear to be due in large part to the introduction and range expansion of nonnative fishes more than overall degradation of habitat quality (Zuckerman 1983, Zuckerman and Langlois 1990, Propst 1999, Calamusso unpublished data). The latter may decrease population persistence through decreased habitat quality and decreased spawning success. Future conservation strategies should consider protection of tributary Rio Grande chub populations from intrusion by nonnative cyprinids, centrarchids, and salmonids, particularly brown trout. Many barriers currently protecting Rio Grande chub (and Rio Grande sucker) populations are unstable beaver dams, debris jams, and water diversions. As with Rio Grande sucker, Rio Grande chub should be introduced to suitable habitat during restoration activities for Rio Grande cutthroat trout. Restoration efforts should be conducted on a watershed scale to ensure stream connectivity and enhance opportunities for metapopulation development.

Spikedace (*Meda fulgida*)

Distribution and Population Trends—The spikedace is endemic to the Gila River Basin of New Mexico and Arizona (Koster 1957, Minckley 1973) and possibly northern Sonora (Propst 1999). Historically, it was common in the Gila River drainage of Arizona downstream to Tempe. It was also found in the Agua Fria, upper Verde, and Salt Rivers (Barber and Minckley 1966, Minckley and Deacon 1968, Minckley 1973). In New Mexico, spikedace were common in the San Francisco River, Gila River, and the lower reaches of the three forks of the Gila River (Anderson 1978, Propst and others 1986). Today, the range of the



Spikedace. (Copyright John Rinne. Used with permission.)

spikedace has been severely reduced (Propst 1999). In Arizona it now is found only in the upper Verde River, Aravaipa Creek, and Eagle Creek (Barber and Minckley 1966, Minckley 1973, Marsh and others 1991). In New Mexico, spikedace have been extirpated in the San Francisco River (Anderson 1978, Propst and others 1986). Propst (1999) reported the distribution of spikedace in the Gila River mainstem as sporadic. The species is found in low numbers in the East Fork of the Gila River and regularly collected in the West Fork of the Gila River, although it appears to be in decline. Spikedace may be extirpated from the Middle Fork of the Gila River (Propst and others 1986, Propst 1999). The species is common to abundant in the upper Gila River in the Gila–Cliff Valley (Propst 1986, Rinne 1999). Spikedace is listed by New Mexico and Arizona as threatened. It is Federally listed as Federal Threatened 31 July 1986.

Preferred Habitat—Spikedace is a small (50 to 60 mm adult size) pelagic cyprinid species that uses moderately deep, low gradient riffles and runs with gradient ranging from 0.3 to 0.5 percent in permanently flowing streams (Sublette and others 1990, Rinne 1991, 1999b). They are associated with sand and gravel substrates (Rinne 1991). Turner and Tafanelli (1983) indicated that velocities most often used by spikedace ranged from 40 to 82 cm/sec with preferred depths ranging from 15 to 18.0 cm. Larvae use low velocity habitats near stream margins with stream depth typically less than 32.0 cm. Rinne and Kroeger (1988) reported spikedace using velocities more than 25 cm/sec with mean water depths of 27 cm.

Habitat use varies geographically and temporally for spikedace along with variation due to age of individuals (Propst and others 1986, Propst 1999). The species is insectivorous with its principal diet composed of mayflies, midges, and caddis flies along with taking other aquatic and terrestrial insects opportunistically (Barber and Minckley 1983).

Breeding Biology—The breeding season for spikedace occurs from February to March at elevations less than 1,000 m, and from April to June at elevations greater than 1,500 m (Anderson 1978, Propst and Bestgen 1991). At the onset of spawning, females move into riffle areas attended by several males. Spawning

takes place in riffles with moderate velocity (25 to 50 cm/sec) over small cobble/gravel substrate (Barber and others 1970, Propst and Bestgen 1991). Eggs of the species are demersal and are broadcast over the substrate, dropping into the interstitial spaces (Anderson 1978) where they incubate for about 4 to 7 days (Snyder 1981). Fecundity is related to age and size. Eggs produced by female spikedace can range from 100 to 300 mature ova (Minckley 1973, Propst and others 1986). Propst and others (1986) reports one age 2 individual 55 mm SL female containing 319 mature ova, and age 1 females averaging 101 mature ova. Spikedace are 5 to 7 mm TL upon emergence and grow at a rate of 1 mm per day (Propst and others 1986).

Management Issues and Recommendations—

The decline in spikedace distribution and abundance is related to the alteration of aquatic and riparian habitats and the introduction of nonnative fishes (Propst 1999). Alteration of habitat for this species includes: ranching and farming practices that have removed instream flow; increased channelization; and increased siltation caused by grazing, road building, and recreational activities; construction of dams; and the introduction of nonnative fishes. The latter both preys upon and competes with spikedace for resources (Sublette and others 1990, Propst 1999). Management practices that address these vectors of decline are needed to ensure the persistence of spikedace. Currently, the USDI Fish and Wildlife Service has designated reaches in the Southwest as designated critical habitat.

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