

Chapter 9

Fish and Aquatic Organisms

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Introduction

The UVR of central Arizona, from its source at Sullivan Lake to the mouth of Sycamore Creek, 60 km (38 mi) downstream, is rare among the State's rivers because it still retains some of its native fish fauna. In 1994, six of the native fishes that were historically recorded in this reach of the Verde still occurred, along with at least seven nonnative species, and many other native species have been incidentally reported (Stefferd and Rinne 1995). The native fish fauna includes longfin dace (*Agosia chrysogaster*), speckled dace (*Rhinichthys osculus*), roundtail chub (*Gila robusta*), spinedace (*Meda fulgida*), desert sucker (*Catostomus clarki*), and Sonora sucker (*Catostomus insignis*) (figs. 9.1 to 9.6; Minckley 1973). The skeletal remains of razorback sucker (*Xyrauchen texanus*) and Colorado pikeminnow (*Ptychocheilus lucius*) have been found at an archaeological site near Perkinsville dated circa 1300 to 1400 A.D. (Minckley and Alger 1968). Both species have recently been stocked in this reach (Hendrickson 1993; Jahrke and Clark 1999). Nonnative species commonly found there include red shiner (*Cyprinella lutrensis*), western mosquitofish (*Gambusia affinis*), fathead minnow (*Pimephales promelas*), common carp (*Cyprinus carpio*), green sunfish (*Lepomis cyanellus*), smallmouth bass (*Micropterus dolomieu*), and yellow bullhead (*Ameiurus natalis*) (Minckley



Figure 9.1—Longfin Dace (*Agosia chrysogaster*). (Photo by John N. Rinne.)



Figure 9.2—Speckled Dace (*Rhinichthys osculus*). (Photo by John N. Rinne.)



Figure 9.3—Roundtail Chub (*Gila robusta*). (Photo by John N. Rinne.)



Figure 9.4—Spikedace (*Meda fulgida*) female (above) and male (below), Verde River, Arizona. (Photo by John N. Rinne.)

Figure 9.5—Desert Sucker (*Catostomus clarki*). (Photo by John N. Rinne.)



Figure 9.6—Sonora sucker (*Catostomus insignis*). (Photo by John N. Rinne.)

1973; Hendrickson 1993). Reasons for retention of this suite of the historic native fish fauna are unclear but may be related to the multiple influences of the geomorphic nature of the river, bed load composition, and its relatively unregulated flow that can scour the channel and dramatically change the physical and biological components, which, in turn, affect the dynamics of the fish assemblage (Brouder 2001; Rinne 2005).

Many independent studies and surveys of the fishes and aquatic ecology of the Southwest and UVR were conducted prior to 1994 (Minckley 1979). Results of some of these activities are available in the published literature and in scattered agency reports. However, not one study consistently investigated the long-term (10+ years) interrelationships of the fish assemblage with natural and/or anthropogenic disturbance events at multiple sites (Rinne 1985a). As of 1994, information was lacking on response of fish assemblages in the UVR to disturbance events such as drought, flooding, land use activities, and invasive species onset (Rinne and Rinne 1993). Research and monitoring were needed to document changes in species abundance and fish assemblages, occurrence through time and space, and fish densities relative to aquatic habitat types, hydrology and geomorphology. Lacking was any long-term and consistent monitoring of the fishery at multiple sites in the UVR.

In 1994, following multiple UVR flood events in winter of 1992/1993, RMRS initiated research and monitoring of a 60-km (38-mi) reach of the UVR between Sycamore Creek and Burnt Ranch (Arizona Game and Fish Heritage Property; Stefferud and Rinne 1995). The objective of the study (Rinne 1994) and monitoring was to determine the roles and relative influence of physical (hydrology

and geomorphology) and biological (introduced fish population dynamics, native versus nonnative interactions, and reintroduction of native species) factors in the sustainability of native fishes in time (10 years) and through the space of seven sites over. Research and monitoring were designed to delineate the effects of flow regimes or hydrographs, local and broadscale geomorphology, and nonnative, invasive fishes on the native fish assemblage (Rinne 2005). From 1995 to present, numerous papers have been published on:

- (1) Changes in fish assemblages over the entire Verde River (Rinne and others 1998, 2005a) and comparison with another Southwest desert river (Rinne and others 2005b);
- (2) Fish-habitat relationships (Rinne and Stefferud 1996; Sponholtz and Rinne 1997; Rinne and Deason 2000);
- (3) Status and habitat use of the threatened spinedace (Neary and others 1996; Rinne 1999a);
- (4) Relationships between hydrology and geomorphology and fish assemblages (Rinne and Stefferud 1997; Neary and Rinne 1998, 2001b; Rinne 2001a, 2002; Rinne and Miller 2006; Rinne and others 2005b, 2005c);
- (5) Stream channel and fish relationships (Rinne and Neary 1997; Rinne 2001d);
- (6) Native fish biology (Rinne and others 2002) and native/nonnative fish interactions (Rinne 2001b, 2004, 2005; Rinne and others 2004); and
- (7) Land use impacts on native fishes (Rinne 1998, 1999b, 2001c; Medina and Rinne 2000; Medina and others 2005; Rinne and Miller 2008).

UVR Study Area

The Verde River flows over 300 km (188 mi) from its headwaters in Big Chino Wash on the Prescott National Forest to its confluence with the Salt River northeast of Phoenix (see Chapter 1; figs. 1.1 and 9.7). The 60 km (38 mi) reach of UVR river corridor above Sycamore Creek or Reach 1 (Rinne and others 1998; Rinne 2001a, 2001b, 2001c, 2001d, 2005b, 2005c) is relatively undisturbed by humans and contains no flow-altering dams or major diversions. However, Sullivan Dam (fig. 9.8; figs. 2.14 through 2.18), constructed in the 1950s, has retained massive amounts of bed load materials from input downstream into the Verde River of over a half century. Primary uses of the river corridor are native fish conservation, general recreation, sport fishing, livestock grazing, and very limited irrigation for private land holdings. Current urbanization and that proposed in the headwaters, as well as the ever-increasing demand for water and groundwater mining, will have over arching impacts on these other river uses.

Methods

Methodologies for research and long-term monitoring are outlined in Rinne (1994) and Stefferud and Rinne (1995). Seven long-term monitoring and research sites were located primarily based on access and spatial disposition over the 60 km-long (38 mi-long) study reach (fig. 9.7). The first is located at Burnt Ranch, about 1.3 km (8 mi) downstream of Sullivan Lake and the last is at the confluence of the Verde River and Sycamore Creek. At each monitoring site, 200- to 300-m (656- to

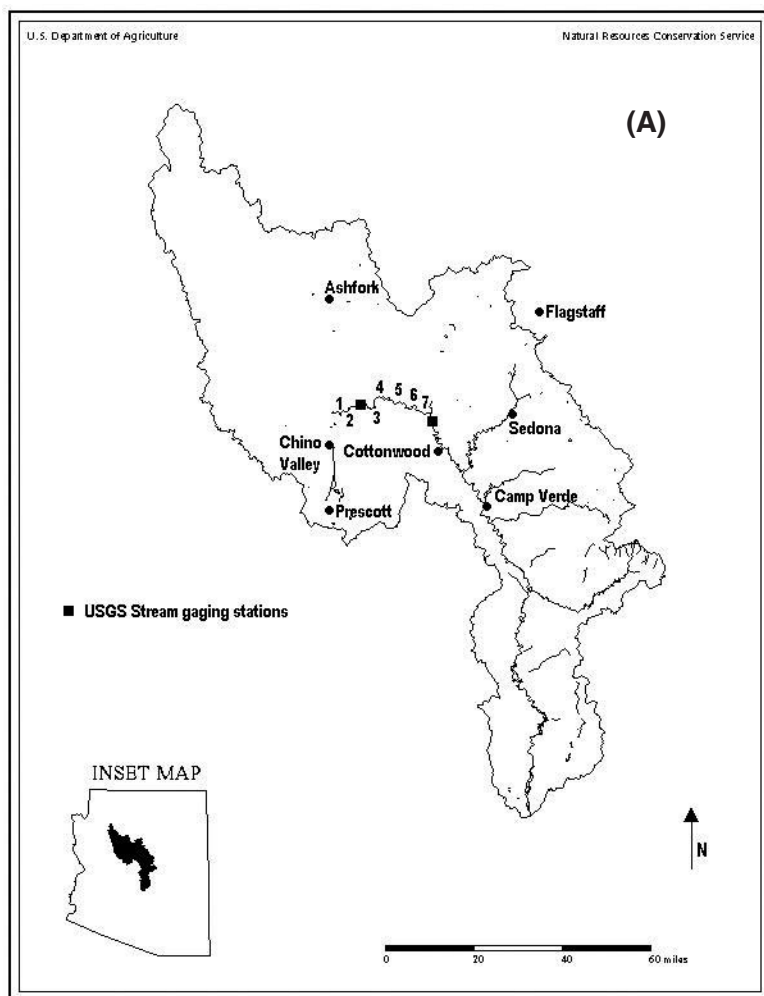


Figure 9.7—(A) UVR long-term fish sampling sites (1-7), local communities, and USDI Geological Survey stream gauge sites (Rinne 2005); and (B) mechanical removal sites at Burnt Ranch and 638 Road.

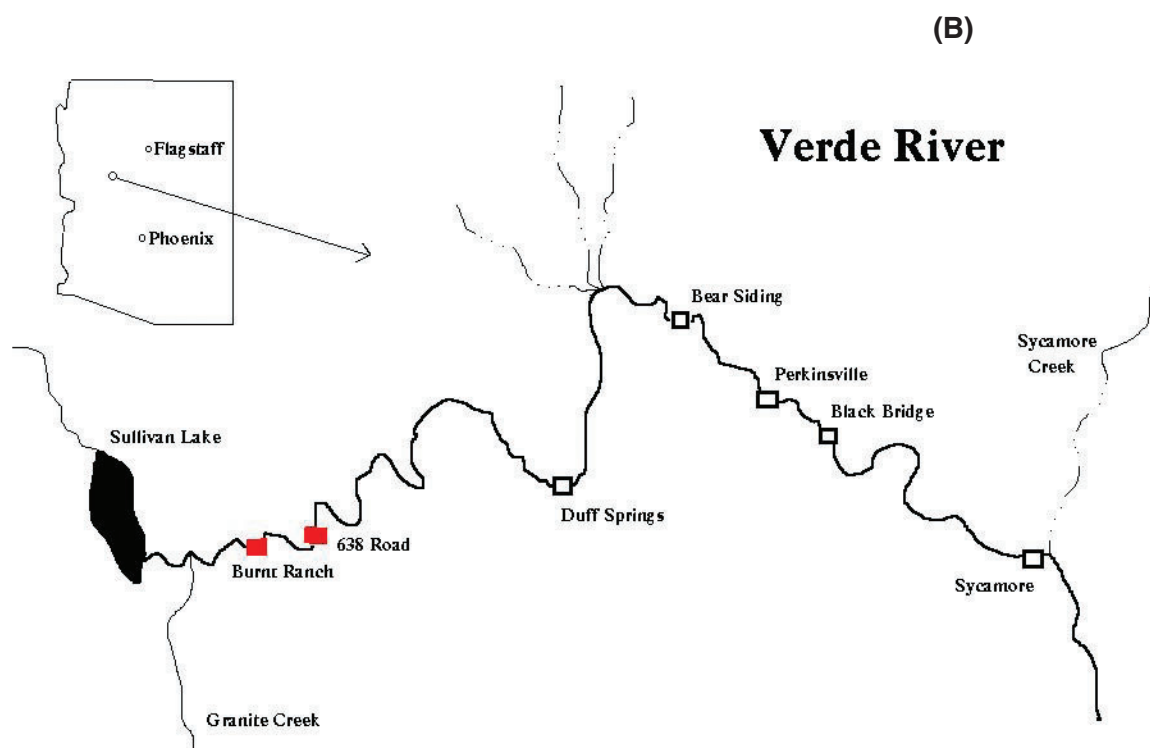




Figure 9.8—Flow over Sullivan Dam during spring runoff, 2005. (Photo by Alvin L. Medina.)

984-ft) reaches of stream were sampled depending on complexity of sites. The overall goal was to encompass the major aquatic habitat types (see the following discussion on macrohabitats) at each monitoring site (Rinne and Stefferud 1996). Habitats were sampled with both backpack, DC electro fishing gear and 3.2-mm (0.13-in) mesh seines that were used actively in glide runs and passively as block nets to capture electro-narcotized fish at the lower end of higher velocity riffles with currents of $>40 \text{ cm s}^{-1}$ (15.7 in s^{-1}) (fig. 9.9). All fish were measured (total lengths in mm), recorded, and returned alive to the stream. In the event of large numbers of individuals of a species in a respective habitat type, only 50 individuals were measured and the remainder were enumerated only. Lengths and widths of each sample reach were measured with metric tape and laser technology, depth was measured with a meter rule, gradients with laser technology, and substrate composition of each sub-reach (i.e., macrohabitat) within a monitoring area with pebble count methodology (Beverger and King 1995). Monitoring sites were consistently maintained over the entire period of study, totaling 15 years as of spring 2008. Hydrologic data are from the USDI Geological Survey Paulden gauge in the upper reaches of the 60 km (38 mi) study area (figs. 9.7 and 9.10).

Mechanical removal of nonnative fishes was initiated at three 1.0 km (0.6 mi) stream reaches in 1999, two in the Burnt Ranch reach of the River and one in the 638 road reach (Site 1, fig. 9.7). This initial, pilot study was sustained until spring 2004 when it was terminated based on results of study and the multiple flooding events occurring in autumn 2004 and winter 2005; fig. 9.11). Methods are in Rinne (2001a, 2001b) and primarily consisted of electro-fishing for capture of fishes within treatment or removal reaches twice a year with single pass methodology. All fishes were measured, natives were returned alive to the stream, and nonnatives were disposed of on-site. Based on results of the original study (Rinne 2001a) and the 2004/2005 flood events that reduced the abundance of nonnative individuals and made marked changes in stream habitat, a modified study was instituted in 2006 and is ongoing (Rinne and others 2006a, 2006b).



Figure 9.9—(A) High-gradient riffle on the UVR downstream of Verde River Ranch, and (B) seining with mesh nets. (Photos by Alvin L. Medina.)



Figure 9.10—U.S. Geological Survey Paulden Gauging Station upstream of the Verde River Ranch near Paulden, Arizona. (Photo by Daniel G. Neary.)



Figure 9.11—2004 flood on the UVR at Perkinsville, Arizona. (Photo by Alvin L. Medina.)

Changes in Fish Assemblages 1994 to 2008

Fish Assemblage Changes

Fish assemblages in the study reach measured by Rinne (2001a, 2005; fig. 9.7) changed markedly over the 15-year period (fig. 9.12). For the first three years following marked, multiple increased instantaneous peak flow events in winter 1992/1993 (fig. 9.13), native species comprised over 80% of the total fish assemblage numbers. During baseflow, drought years between 1997 and 2004, fish assemblages were similar, but inverse, dominated by nonnative fish species that accounted for about 70 to 85% of the total fish assemblage at the seven long-term research/monitoring sites. All native species, including the threatened spiketail, declined substantially in samples during baseflow, drought years (Rinne 2005). All three smaller-sized species—speckled dace, longfin dace, and the threatened

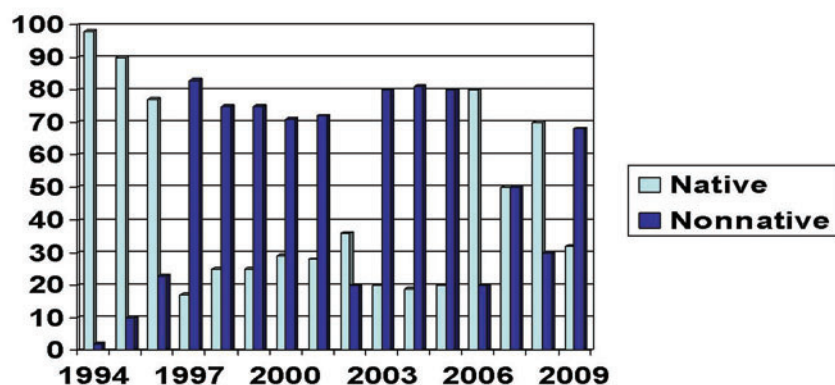


Figure 9.12—Changes in UVR native and nonnative fish assemblages (percentages), 1994 to 2009.

MAXIMUM IP FLOW EVENTS, UPPER VERDE RIVER, 1974-2008

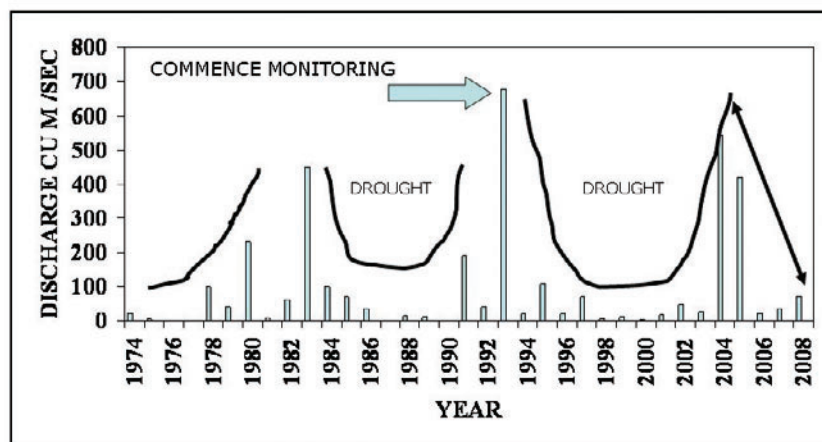


Figure 9.13—Maximum instantaneous peakflow in $\text{m}^3 \text{sec}^{-1}$ measured at the Paulden gauge on the UVR, 1974 to 2008.

spikedace—became virtually absent. The spikedace has been absent from samples at all seven long-term sites since 1997 (Rinne 1999a). Although the three larger-sized, longer-lived species—desert sucker, Sonora sucker, and roundtail chub—were yet present, their numbers were likewise markedly reduced by 1996 (Rinne and others 2005c), and recruitment of all three of these species was reduced to absent (figs. 9.14A and 9.14B).

By contrast, nonnative species such as smallmouth bass, green sunfish, and red shiner oscillated in abundance from 1994 to 2008. The length of native species dominance after flood events differed between the 1992/1993 floods and the 2004/2005 floods. After the former events, native fishes dominated the fish assemblage in the UVR from 1994/1996. However, after the 2004/2005 floods, the total fish assemblage was native-dominated. But, by spring 2009, natives declined to <33% of the total fish assemblage (fig. 9.12).

In summary, fish assemblages responded markedly to flow regimes of the UVR over a 15-year period. However, flows do not appear to be the only operative factor affecting fish assemblages. That is, the response of fish assemblages post-flooding and prior to commencement of monitoring in 1994 was different from assemblages following recent, 2004/2005 flood events. The threatened spikedace, speckled and longfin daces—all short-lived, small-sized (<70 mm or <2.76 in) species—have virtually disappeared from samples in the uppermost reach of this desert river

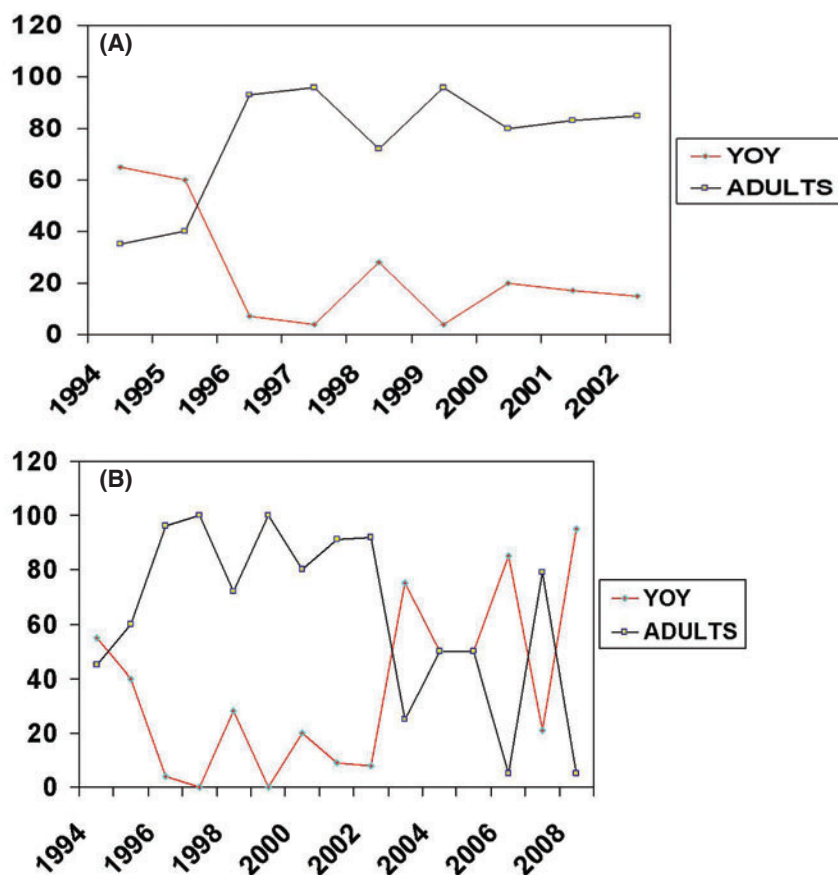


Figure 9.14—(A) Recruitment of sonora sucker in the UVR, 1995 to 2002, and (B) recruitment of roundtail chub in the UVR, 1995 to 2002 (YOY = young-of-the-year).

compared to 1994 samples (figs. 9.15A, 9.15B, and 9.15C). In spring 2006, only speckled dace increased in abundance in samples ($n = 26$). Spikedace and longfin dace were absent. The three larger-sized (>300 mm or >11.8 in), longer-lived (5 to 10 years) species also have become markedly reduced in numbers (figs. 9.16A, 9.16B, and 9.16C). The competitive and predatory influence of nonnative species (Rinne and Minckley 1985; Rinne 1995b) have to be considered equally, or perhaps even more importantly than flow regimes. Most likely, the two factors are interactive and synergistic. There are ongoing efforts described in the following Mechanical Removal section to physically remove nonnative fishes and to compare the response in treated reaches to that at contiguous control reaches at two of the long-term monitoring sites—Burnt Ranch and 638 Road—and at the other five long-term monitoring reaches.

At all seven sites, the native fish component dominated (82 to 86%) in numbers over the nonnative component (table 9.1, fig. 9.12). However, such predominance of native species was variable, comprising 54% of the total fish assemblage in autumn 1994 and increasing to 85% in the sampling reach by spring 1995. By autumn 1995, native species comprised 88% of the total fish assemblage. Similarly, sampling following the winter 1992/spring 1993 flood event, and, synchronous with data collected by RMRS personnel, indicated that natives comprised 96% of the total fish assemblage (Rinne and Stefferud 1997).

Changes in Species Abundance, 1994 to 2008

Numbers of most fish species decreased markedly between spring 1994 and 1995 sampling; only green sunfish increased (table 9.1). Native fish decreased from between 56 and 99% (mean of 83%) and nonnatives decreased by 11 to 100%. Yellow

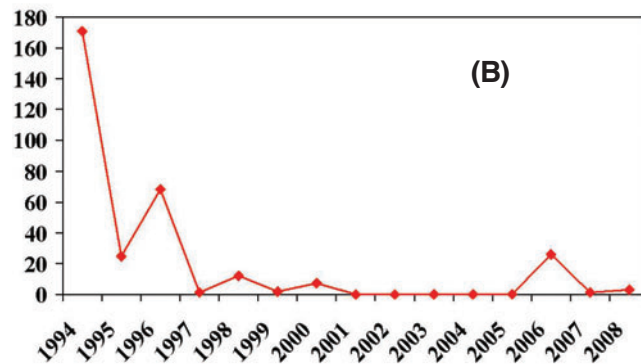
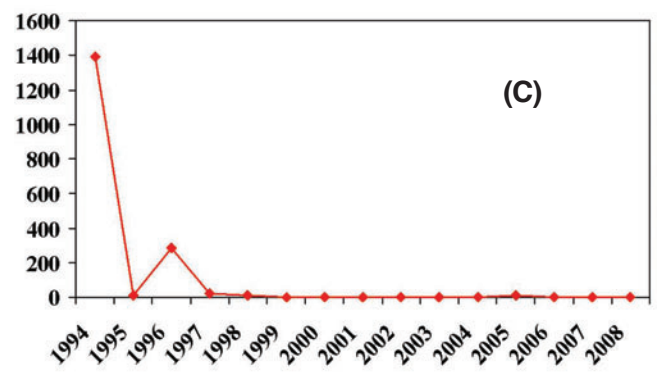
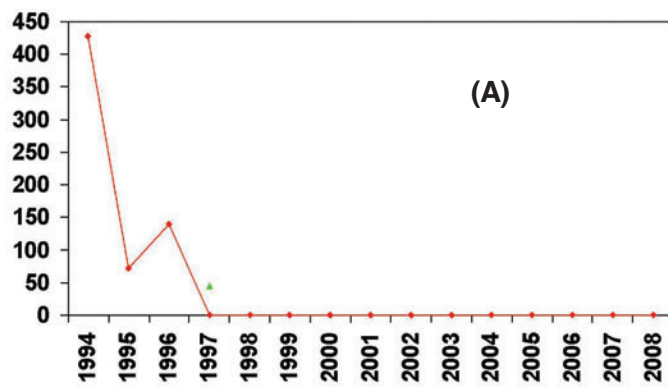


Figure 9.15— Total number of small-sized fish collected in the UVR seven-site sampling areas, 1994 to 2008: (A) spiketail, (B) speckled dace, and (C) longfin dace.

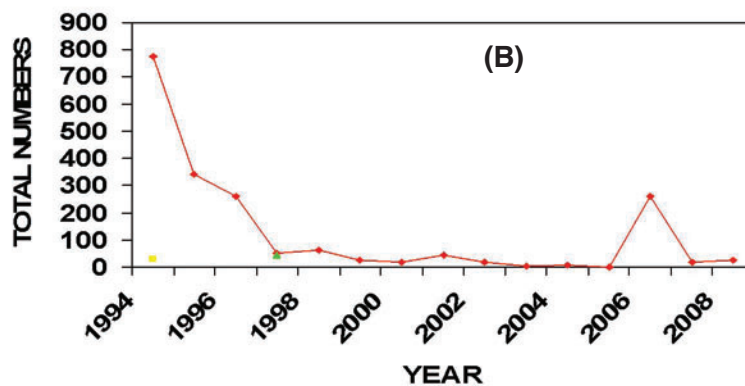
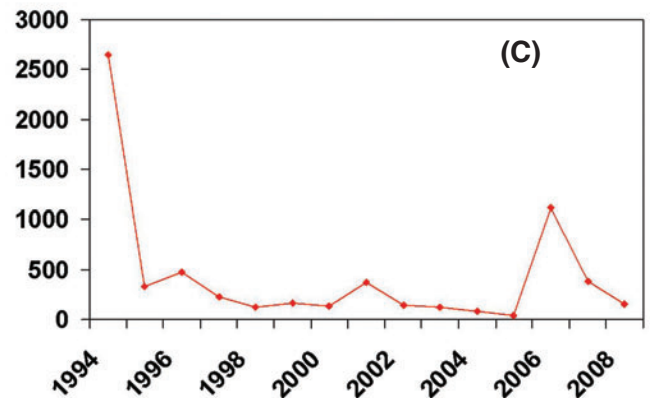
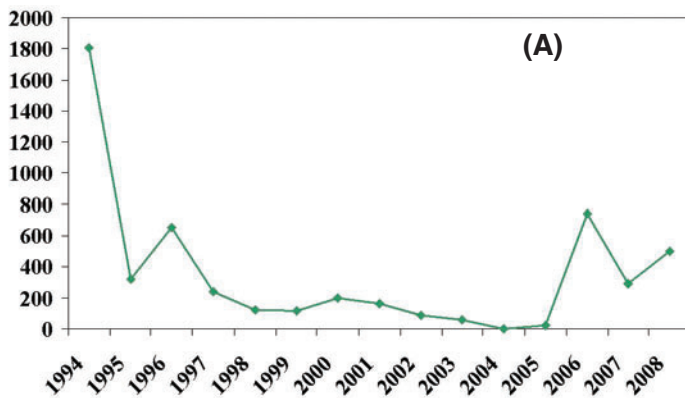


Figure 9.16— Total number of large-sized fish collected in the UVR seven-site sampling areas, 1994 to 2008: (A) sonora sucker, (B) roundtail chub, and (C) desert sucker.

Table 9.1—Fish community dynamics (number and percent change) in the UVR, spring 1994 to 1996. Values in () are percentages of the total fish assemblage, and bold numbers are total numbers of sampled fish in the native and nonnative components. Species abbreviations are shown and are the same for tables 9.2 and 9.3.

Species	Species Abbreviation	Spring 1994 Fish	Spring 1995 Fish	Change ¹	Spring 1996 Fish	Change ²
		<i>Numbers</i>	<i>Numbers</i>	<i>%</i>	<i>Numbers</i>	<i>%</i>
Native						
Desert Sucker	CACL	2644	247	-91	471	+91
Sonoran Sucker	CAIN	1810	322	-82	654	+103
Roundtail Chub	GIRO	776	341	-56	259	-24
Longfin Dace	AGCH	1319	12	-99	282	+2250
Spikedace	MEFU	428	72	-83	140	+94
Speckled Dace	RHOS	171	25	-85	68	+172
TOTAL		7148	1019 (85%)	-861	1874 (85%)	+462
Introduced						
Red Shiner	CYLU	1473	97	-93	275	+183
Common Carp	CYCA	23	6	-74	13	+117
Smallmouth Bass	MIDO	14	10	-29	32	+220
Green Sunfish	LECY	5	29	+580	6	-79
Yellow Bullhead	AMNA	36	32	-11	6	-81
Fathead Minnow	PIPR	6	0	-100	0	0
TOTAL		1558	174 (15%)	-891	332 (15%)	+912

¹ % change 1994 to 1995

² % change 1995 to 1996

bullhead was reduced by only 11% and smallmouth bass by 29%. Red shiner decreased by 93%, and no fathead minnows were found in spring 1995.

Biannual sampling at the upper- and lower-most sites in the river offers a more detailed picture of population dynamics (table 9.2). All native species, except roundtail chub, declined from spring 1994 to autumn 1994 at the Burnt Ranch site, decreased further by spring 1995, and increased in autumn 1995 and spring 1996 following flooding in spring 1995. Most nonnative species varied little between samples. The introduced red shiner population at Burnt Ranch held stable between spring and autumn 1994 and then declined markedly (>80%) by spring 1995 only to then increase markedly (95%) by autumn 1995. By comparison, at the Sycamore site, red shiner increased more than 100-fold between spring and autumn 1994 only to collapse (-99%) to very low population levels in 1995 and 1996. At Sycamore, common carp numbers were low from spring 1994 to spring 1996. Similarly, smallmouth bass numbers remained low during all sample periods, disappeared at Sycamore in spring 1995, and increased slightly by autumn 1995. Green sunfish were absent at these two sites in 1994, but were present and increasing at the Burnt Ranch site by autumn 1995. Numbers of yellow bullhead were generally low and declined steadily from spring 1994 to spring 1996.

Mechanical Removal: 1999 to 2003 and 2006 to 2009

1999 to 2003 Pilot Study

The marked decline of the native fish assemblage in the UVR mimics that presented for other stream and rivers in the generally reported negative impact of nonnative fishes on native fishes (Minckley and Deacon 1991; Rinne and Minckley

Table 9.2—Seasonal fish assemblage dynamics at Burnt Ranch (BR) and Sycamore Creek (SC), UVR, 1994 to 1996: (A) Spring 1994, Autumn 1994, and Spring 1995; and (B) Autumn 1995 and Spring 1996.

A									
	Spring 1994			Autumn 1994			Spring 1995		
Species	BR	SC	Total	BR	SC	Total	BR	SC	Total
	Numbers			Numbers			Numbers		
Native Species									
CACL	339	379	718	31	93	124	15	29	44
CAIN	278	223	501	214	25	239	60	37	97
GIRO	15	165	180	50	17	67	3	104	107
AGCH	1072	1	1073	94	0	94	0	0	0
MEFU	257	92	349	93	0	93	33	17	50
RHOS	0	19	19	0	1	1	0	0	0
Nonnative Species									
Cylu	39	3	42	50	395	445	7	5	12
Cyca	1	4	5	67	1	68	0	2	2
Mido	2	3	5	2	3	5	3	0	3
Lecy	0	0	0	0	0	0	0	2	2
Amna	2	10	12	1	6	7	2	1	3
PIPR	0	0	0	5	0	5	0	0	0

B									
	Autumn 1995			Spring 1996					
Species	BR	SC	Total	BR	SC	Total			
	Numbers			Numbers					
Native Species									
CACL	44	77	121	79	38	117			
CAIN	103	93	196	92	41	133			
GIRO	40	6	46	23	25	48			
AGCH	397	0	397	91	1	92			
MEFU	290	0	290	33	51	84			
RHOS	0	12	12	0	0	0			
Introduced Species									
Cylu	151	7	158	88	9	97			
Cyca	3	0	3	1	1	2			
Mido	0	9	9	5	0	5			
Lecy	7	0	7	1	0	1			
Amna	5	8	13	1	0	1			
PIPR	0	0	0	0	0	0			

1991; Rinne 2003a) and suggests that nonnative species of fishes negatively, and often dramatically, impact native species of fishes. Such disturbing patterns in the UVR, when combined with the literature on the generally reported negative impact of nonnative fishes on native fishes in the western United States, infers the likely negative impact of both large (smallmouth bass, green sunfish, and yellow bullhead) and small (western mosquitofish and red shiner) fishes on young-of-the-year (YOY) native fishes (Carpenter and Mueller 2008). Because of this interaction, it was determined in autumn 1999 to treat several reaches of the UVR by mechanically removing nonnatives to determine the response of native species.

Nonnative species of fishes have been intensively and extensively introduced throughout the western United States over the past century (Minckley and Deacon 1991; Rinne 2003a; Rinne and others 2005a). Establishment of nonnative fishes assemblages has been a major impediment to the conservation and sustainability of native fishes in rivers and streams in the southwestern United States. A common

method of removing nonnative species has been chemical treatment with piscicides to renovate streams to sustain and enhance native fish species (Rinne and Turner 1991; Finlayson and others 2000). However, this management tool is increasingly scrutinized by the public and often is subject to administrative delays. Further, the politics of managing sport fishes alongside native fishes is complex, usually contradictory, and most likely mutually exclusive (Rinne and others 2004; Marsh and Pacey 2003; Clarkson and others 2005).

Mechanical removal has recently been considered as an alternative tool to reduce/suppress nonnative fish species and restore native fish assemblages as it has increased in popularity recently. In some regions of the United States, substantial funds are being expended to implement these programs (Mueller 2005). The question becomes, “Is this approach a viable tool for the conservation and sustainability of native fishes?” Further questions that managers need answers to are:

- (1) Does mechanical removal reduce nonnatives sufficiently to ensure sustainability of native populations?
- 2) What are the thresholds at which native species commence to respond to this treatment?
- (3) What are the problems and probabilities of success, and how do politics and the public influence these probabilities?
- (4) Does effectiveness relate to methods and approaches, species-specific response, stream size, and temporal-spatial aspects?
- (5) Are these more often only “feel good” projects that expend considerable fiscal and human resources and really buy resource agencies little toward restoring and sustaining native fish assemblages?

Pilot Study: Mechanical Removal of Invasive Fishes

In 1999, because of the sustained loss of the native component of the fish assemblage, a pilot nonnative, mechanical fish removal project was initiated in the extreme UVR. The treatment involved removal of invasive fish species from three reaches of the river between September 1999 and February 2005. The response of the native fish component was estimated based on presence and recruitment of YOY and Age 1 individuals. Flooding that commenced in autumn 2004 and continued through winter of 2005 provided an opportunity to address some of the problems encountered in the pilot study (see the “Results” section).

Three experimental reaches, each approximately 1 km (0.6 mi) in length, were selected to be treated in the upper most reaches of the river. Two were established in the Burnt Ranch study area and one was placed at the 638 Road long-term study reach (fig. 9.7B). Commencing in autumn 1999 and ending in spring 2004, these sites were sampled in spring and autumn with a single pass approach and backpack DC electro-fishing units. All fishes were enumerated, and natives were returned alive to the stream. Captured nonnative fish were removed from the river. In sampling, YOY and age 1 native fishes were captured, measured, recorded, and returned alive to the stream. The relative numbers and percentages of these age class individuals were then compared to the percentage composition recorded at the Burnt Ranch and 638 Road long-term sampling sites to determine if a positive response in native fish recruitment resulted from mechanical removal of predatory fishes (Rinne 2001b). In June 2006, a modified approach was adopted that encompassed almost 8 km (5 mi) of the UVR. Previously treated reaches and long-term monitoring sites at Burnt Ranch and 638 Road were included in the study.

Results of the Pilot Study

Almost 4,000 invasive, nonnative potentially predatory fishes were removed mechanically from the river in the initial treatment period (fig. 9.17). In contrast to expectations, numbers of nonnative individuals removed increased with successive treatments. This was especially true at the Burnt Ranch sites. Overall, smallmouth bass were removed in the greatest numbers (fig. 9.18). Total numbers of smallmouth bass increased from 1999 to 2003 (fig. 9.19), but mean size of this species decreased from 160 to 180 mm (6.3 to 7.1 in) to about 120 mm (4.7 in) during that same time period (fig. 9.20).

Criteria for success of removal were based on comparative numbers of YOY and Age 1 of the three larger-sized native species (figs. 9.14A and 9.14B). The most positive response was at the 638 Road treatment reach (figs. 9.21A and 9.21B). There was no consistent, significantly positive response of YOY and Age 1 natives at removal reaches versus those same age classes at non-treated, long term monitoring sites.

Results of four years of Pilot Study removal reported by Rinne (2001b) indicated that:

- (1) Native fishes did not respond consistently or positively.
- (2) Nonnative species, especially smallmouth bass, increased in numbers although they were reduced in size.
- (3) The single pass removal was not effective.

Conclusions from the study were:

- (1) Intensity of single pass removal, once a year, was not sufficient to adequately reduce nonnative fish species. A three-pass system requires more effort but it is much more effective.
- (2) Extensive streambank and in-stream vegetation reduced efficiency of removal.
- (3) The limited spatial extent of 1-km (0.6-mi) treatment reaches did not preclude immigration into sites.
- (4) Vegetation and increases in invasive fishes, combined with the marked reduction of all native species prior to study, precluded effective removal of nonnative species.

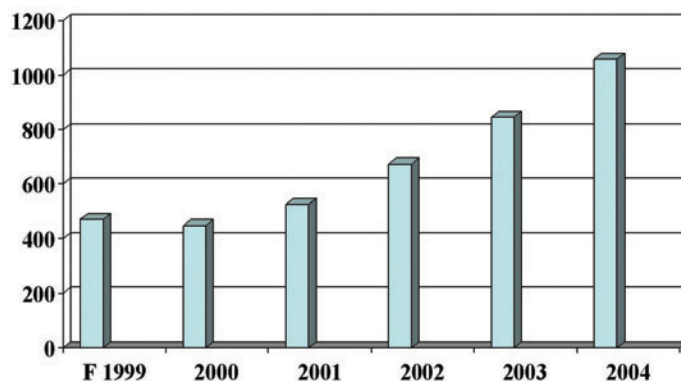


Figure 9.17— Total number of non-native fish removed by year during the fish removal study, UVR, 1999 to 2004.

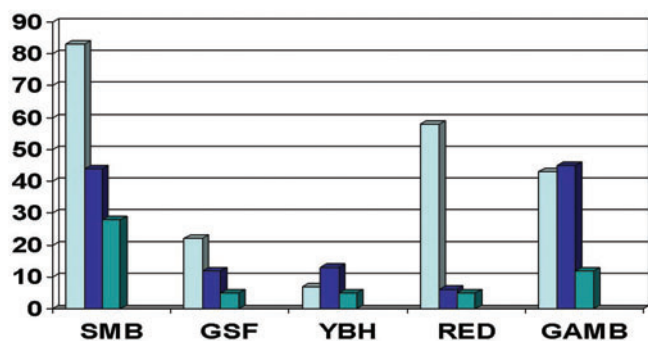


Figure 9.18—A one-time three-pass depletion on non-native fish species in the UVR in 2004 (SMB = smallmouth bass, GSF = green sunfish, YBH = yellow bullhead, RED = red shiners, and GAMB = western mosquitofish).

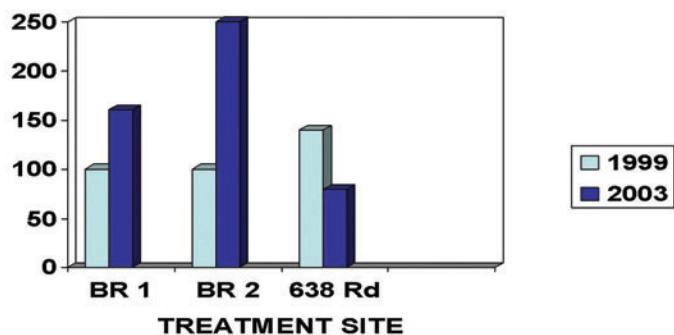


Figure 9.19—Change in total number of smallmouth bass at three treatment sites in the UVR, 1999 to 2003.

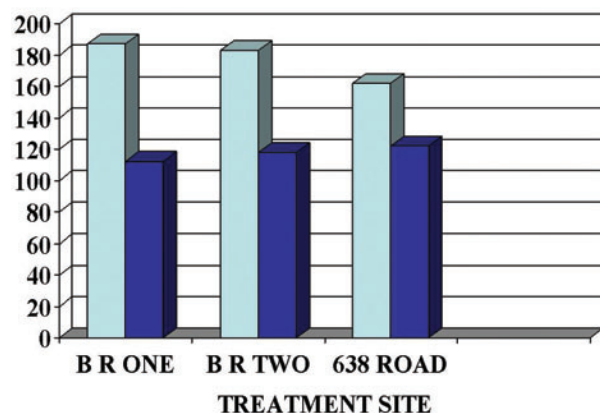


Figure 9.20—Change in mean size (mm) of smallmouth bass at Burnt Ranch and 638 Road, 1999 and 2003.

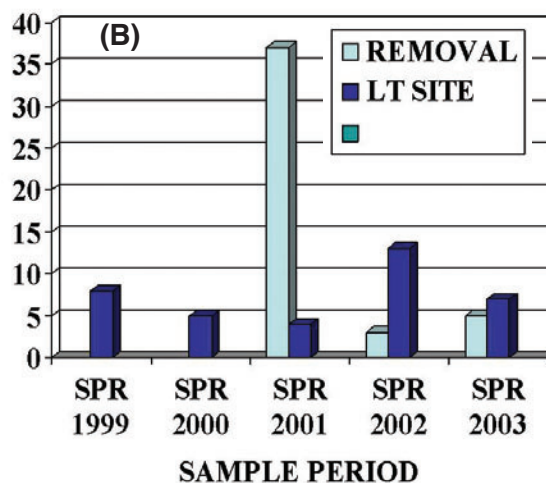
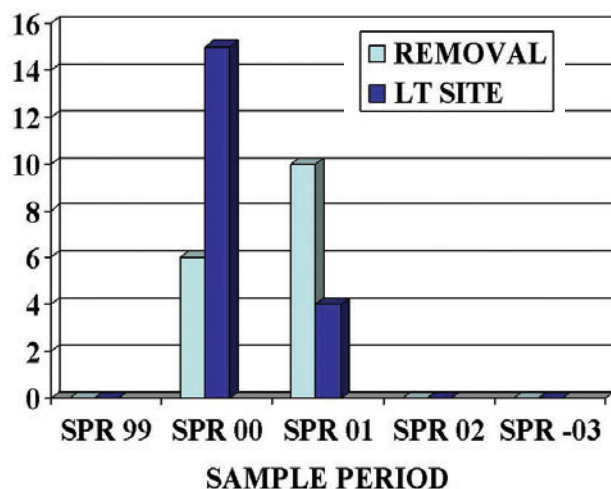


Figure 9.21—Percentage removals at removal study sites versus 638 Road long-term site, 1999 to 2003: (A) roundtail chub and, (B) sonora sucker.

Rationale for Lack of Success of Pilot Mechanical Removal

When studying natural systems, multiple, interactive factors are basic and either may be either antagonistic or synergistic. Both natural and anthropogenic activities were simultaneously occurring on the UVR and appear to have confounded and precluded achieving desired goals of the removal exercise. In summary, the four years of mechanical removal of nonnative fishes failed to invoke a consistent favorable response by the three larger-sized, longer-lived species. The conclusions here, based on removal between 1999 and 2004, are:

- (1) Single pass electrofishing can remove only an average of 50% or less of the fish.
- (2) Treatment twice a year was not sufficient to suppress nonnatives.
- (3) Streambank and instream vegetation increase that was a result of livestock grazing cessation reduced efficiency of removal (see figs. 2.45 and 2.46).
- (4) Populations of the three larger-sized native indicator species were reduced below a threshold (figs. 9.16A, 9.16B, and 9.16C) from which they could only slowly recover.

Further, these and other confounding factors potentially can, and likely did, have an influence on the outcome of this study:

- (1) Livestock removal, in turn, resulted in denser riparian vegetation and increased hiding and avoidance cover during removal.
- (2) Antagonistic management activities in the form of road closures and removal of sport fishing limits that reasonably reduce angling, even when anglers can take a greater number of nonnative, sport species, reduce depredation of nonnative fishes.
- (3) Sustained drought and base flow regimes in absence of significant ($>141.6 \text{ m}^3 \text{ s}^{-1}$ or $>5,000 \text{ ft}^3 \text{ s}^{-1}$) instantaneous peak flow events favored nonnative fishes (fig. 9.22).

No attempt was made to control the highly probable immigration of fishes into and out of treated reaches. Extensive and intensive flow events in winter 2004/2005 dramatically altered and reset in-stream and near stream habitat, and provided a window of opportunity to more effectively mechanically remove nonnative fishes from the UVR.

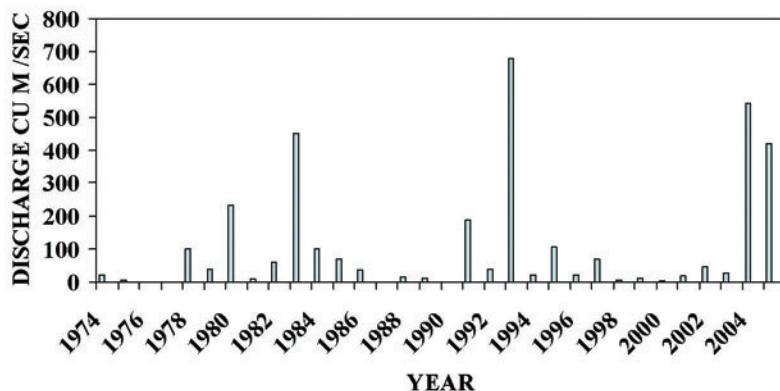


Figure 9.22—Maximum instantaneous peakflow events in $\text{m}^3 \text{ sec}^{-1}$ measured at the Paulden gauge, UVR, 1974 to 2005.

2006 to 2009 Modified, Ongoing Mechanical Removal

On September 19, 2004, the Verde River at the Paulden gage went from base flow of about $0.6 \text{ m}^3 \text{ sec}^{-1}$ ($20 \text{ ft}^3 \text{ sec}^{-1}$) to over $500 \text{ m}^3 \text{ sec}^{-1}$ ($17,657 \text{ ft}^3 \text{ sec}^{-1}$) (fig. 9.22) within hours as a result of heavy precipitation (10+ cm or 4+ in) in the upper Chino watershed (flow similar to fig. 9.8 and actually fig. 9.11). This event provided an opportunity to test the hypothesis of importance of the flood or the instantaneous peak flow component of the hydrograph on fish assemblages. More importantly, it presented a window of opportunity immediately post-flooding to further mechanically suppress the invasive component of the fish assemblages at both a larger and more intensive scale than previously possible (Rinne 2001a). Flooding reduced both the density of nonnatives and the in-stream and stream bank cover for the invasive species (e.g., smallmouth bass, green sunfish, and mosquitofish) which facilitated more effective, continued suppression of the nonnative fishes. Accordingly, with the likelihood of increased post-flood spawning and recruitment by native species in spring-summer 2005/2006, a more spatially and temporally robust experimental program of removal was launched that addressed apparent shortcomings and problems encountered in the initial 1999/2004 treatment efforts.

The on-going experimental effort started in April 2006 was comprised of eight 1.0 km (0.6 mi) contiguous units (fig. 9.23). The study area encompassed the three previously (1999 to 2004) treated long-term monitoring sites (Burnt Ranch and 638 Road) and was to be sampled two to three times a year (spring, summer, and autumn). One “frame of reference” for analyzing the effects of the mechanical removal treatment is the long term monitoring dataset at Burnt Ranch and 638 Road (figs 9.7A and 9.7 B; figs. 9.24A, 9.24B, and 9.24C).

Results of Post-Flood Treatments

The first post-flood treatment occurred from June 26 to 28, 2006, and indicated that native fishes were yet the major component of the fish assemblage (table 9.3). Desert sucker were most abundant, Sonora sucker ranked second in abundance, and roundtail chub were least abundant (table 9.3). Of the three smaller-sized

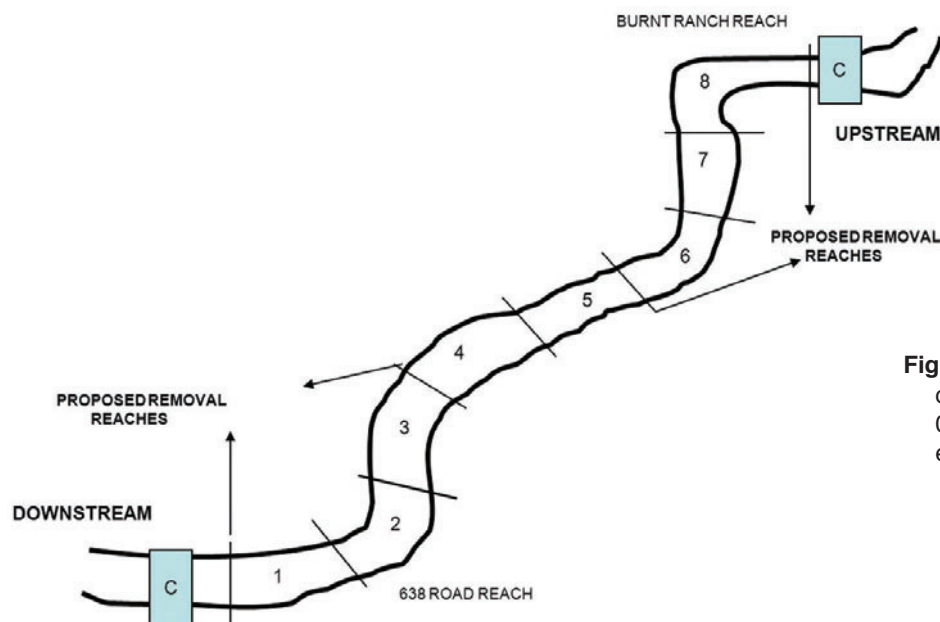


Figure 9.23—Diagrammatic illustration of positioning controls (“C”) and 0.5 km (0.3 mi) sub-sample reaches established in summer 2006.

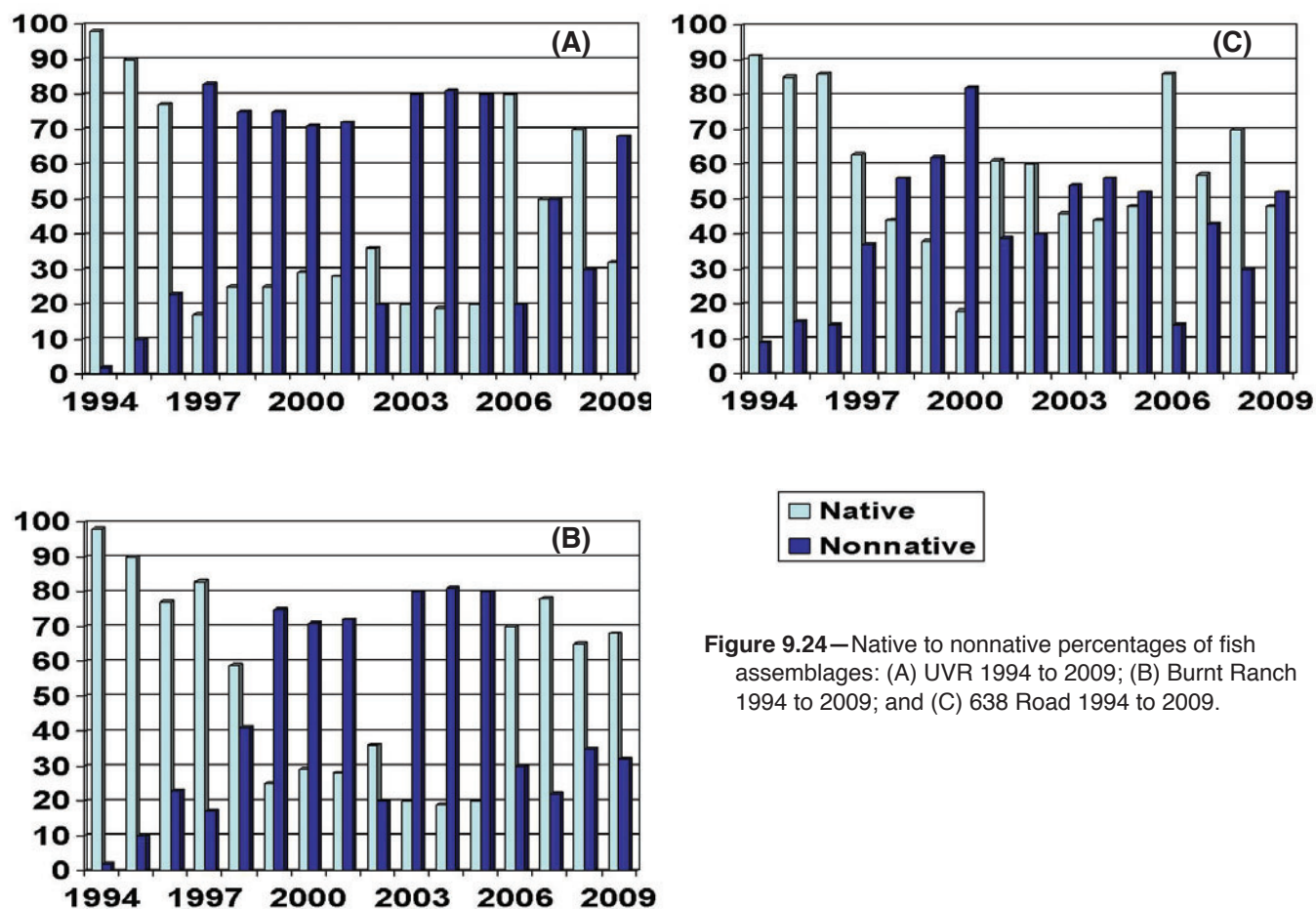


Figure 9.24—Native to nonnative percentages of fish assemblages: (A) UVR 1994 to 2009; (B) Burnt Ranch 1994 to 2009; and (C) 638 Road 1994 to 2009.

Table 9.3—Relative composition of fish species in the eight mechanical removal reaches, UVR, June 2006.

Species	Reach							
	1	2	3	4	5	6	7	8
	<i>Number</i>							
Native Species								
Longfin dace	0	0	0	1	0	0	0	0
Desert sucker	409	247	302	163	215	305	333	62
Sonora sucker	450	214	159	352	308	236	294	75
Roundtail chub	23	19	19	30	21	34	19	18
Introduced Species								
Smallmouth bass	65	17	59	59	85	102	125	44
Green sunfish	1	2	6	0	0	4	0	0
Yellow bullhead	18	13	24	4	18	23	29	7
Red shiner	258	89	254	336	43	57	7	23
Fathead minnow	0	0	2	0	0	4	0	0
TOTALS	1229	612	830	946	690	811	809	229

native species, including the threatened spikedace, only a single longfin dace was encountered during the three-day effort. Additional removal treatments were performed in July 2006, April 2007, July 2007, April and June 2008, and April and June 2009.

Figures 9.25A and 9.25B depict native and nonnative components of the total fish assemblages in spring 2008 after two years of mechanical removal at the 0.5-km (0.3-mi) treatment sites in 2006 and 2007 are shown in figs. 9.25A and 9.25B). In the summer of 2008 (third year of treatment), all eight treated reaches contained 60 to 80% native individuals. By comparison, the lower control reach contained 63% nonnatives (fig. 9.26). These proportions were near the same as those at the eight treated sites and one control site in spring 2006. However, a year later, in April 2007, over half of the treated sites were trending toward a nonnative assemblage and four were still native-dominated (fig. 9.25B). By spring 2008, all treated sites were native-dominated (mostly by 70% or more; fig. 9.26). By contrast, the lower control reach was comprised of over 60% nonnative individuals. Finally, in spring 2009, the native fish component had again declined at three of the four sites in the 638 Road reach (fig. 9.27). All removal sites (R-5 to R-8) in the Burnt Ranch reach sustained native fish dominance, comprising 50 to almost 80% of the estimated total fish assemblage. Sampling two months later in June 2009 revealed a reversion of all sites to a native-dominated assemblage (fig. 9.27). However, the control site at the lower end of the 638 Road treatment reach also was estimated to have over 60% native fishes. By contrast, instituting sampling of an upstream control site (R-9) not previously sampled from 2006 to 2008, indicated over 60%

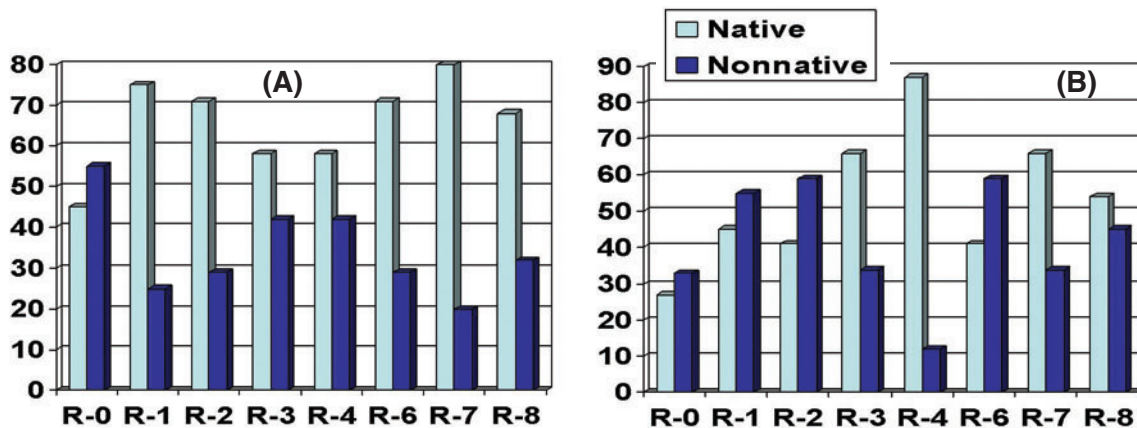


Figure 9.25—(A) Nonnative to native fish percentages at eight mechanical removal reaches (R-1 to R-8) and one control reach (R-0), UVR, 2006; and (B) native to nonnative fish percentages at eight mechanical removal reaches (R-1 to R-8) and one control reach (R-0), UVR, 2007.

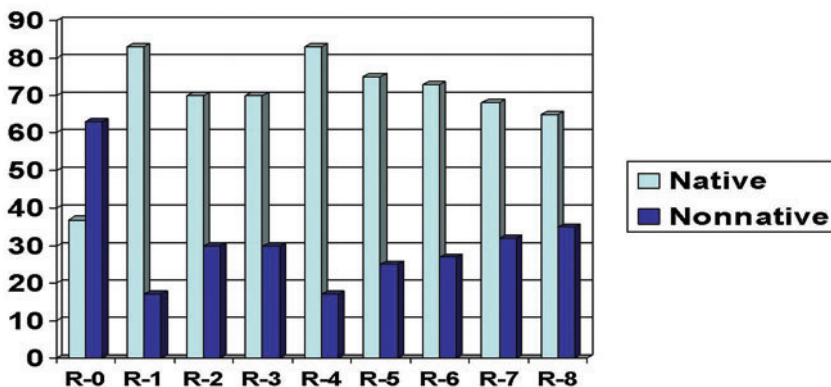


Figure 9.26—Native to nonnative fish percentages at eight mechanical removal reaches (R-1 to R-8) and one control (R-0), UVR, 2008.

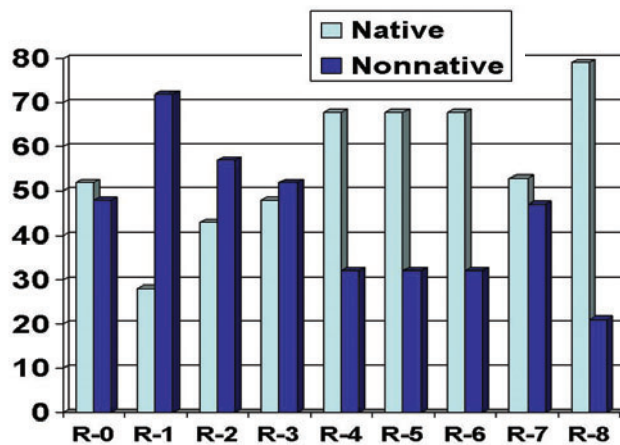


Figure 9.27—Native to nonnative fish percentages at eight mechanical removal reaches (R-1 to R-8) and one control (R-0), UVR, 2009.

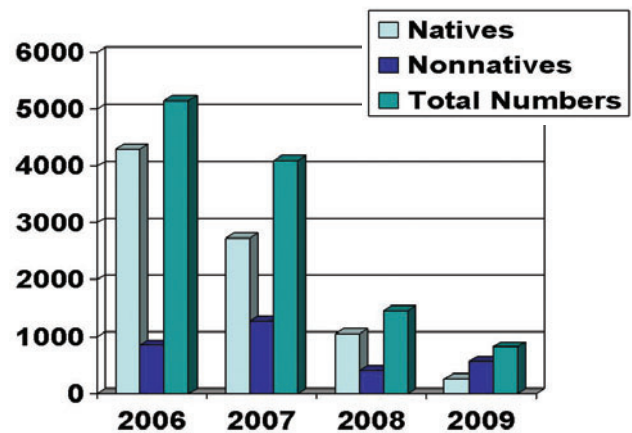


Figure 9.28—Total, native, and nonnative fish numbers in treatment sites, UVR, spring 2006 to 2009.

nonnative component. During this same time period total numbers of both native and nonnative fish declined (fig. 9.28). Of the nonnatives, red shiner numbers were depleted the most by mechanical removal (fig. 9.29).

Results of the most recent sampling in 2008 indicated that the two long-term monitoring sites (Burnt Ranch and 638 Road) within the treatment sites contained higher percentages (70 to 73%) of native fishes than did two contiguous downstream sites, Duff Springs and Bear Siding (17 to 50%; fig. 9.30). In 2009, there was a substantial increase in nonnative species at the Duff Springs, Perkinsville, and Black Bridge sites. Nonnative species ratios at Bear Siding did not change much (fig. 9.31).

Comparative sampling at the seven long-term research/monitoring sites documented a marked, inverse decline in the native component of the fish assemblage from spring 2008 to 2009 (figs. 9.12, 9.30, and 9.31). This decline largely resulted from the continuing declines of the native component at those sites dominated by natives in spring 2008 and the reversal of dominance at the 638 Road, Perkinsville

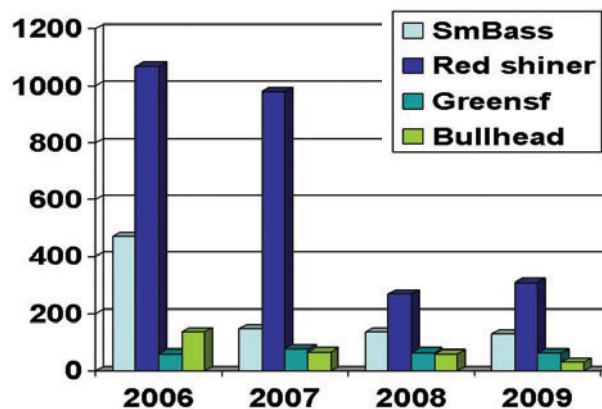


Figure 9.29—Temporal changes in nonnative fish species numbers in mechanical removal sites, UVR, 2006 to 2009.

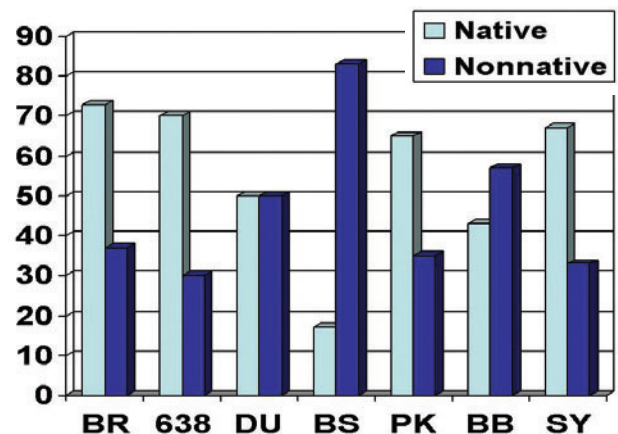


Figure 9.30—Comparative native to nonnative fish percentages at seven long-term sampling sites, UVR, spring 2008. Sites are left to right Burnt Ranch (BR), 638 Road (638), Duff Springs (DU), Bear Siding (BS), Perkinsville (PK), Black Bridge (BB), and Sycamore Canyon (SY).

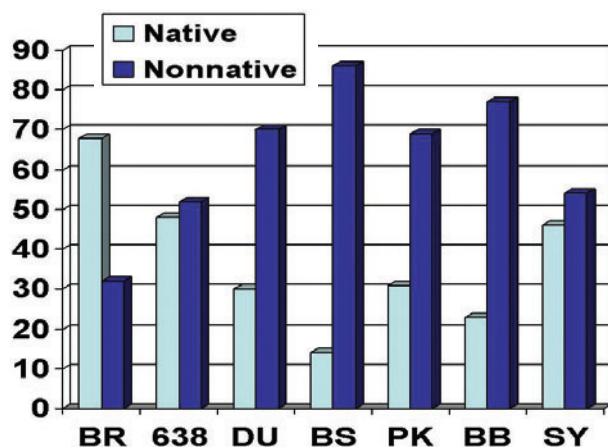


Figure 9.31—Comparative native to nonnative fish percentages at seven long-term sampling sites, UVR, spring 2009. Sites are left to right Burnt Ranch (BR), 638 Road (638), Duff Springs (DU), Bear Siding (BS), Perkinsville (PK), Black Bridge (BB), and Sycamore Canyon (SY).

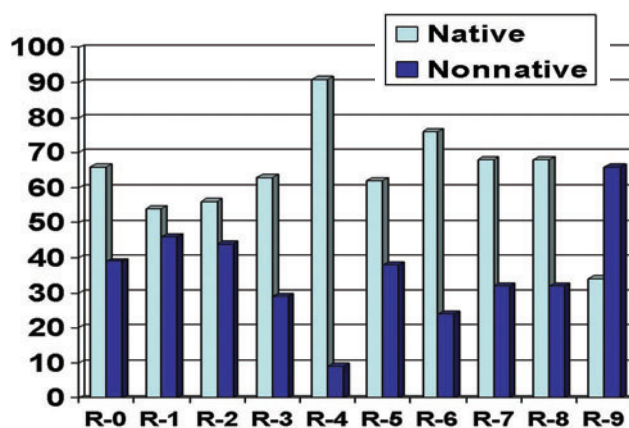


Figure 9.32—Native to nonnative fish percentages at eight mechanical removal reaches (R-1 to R-8) and two control reaches (R-0 and R-9) at Burnt Ranch, UVR, June 2009.

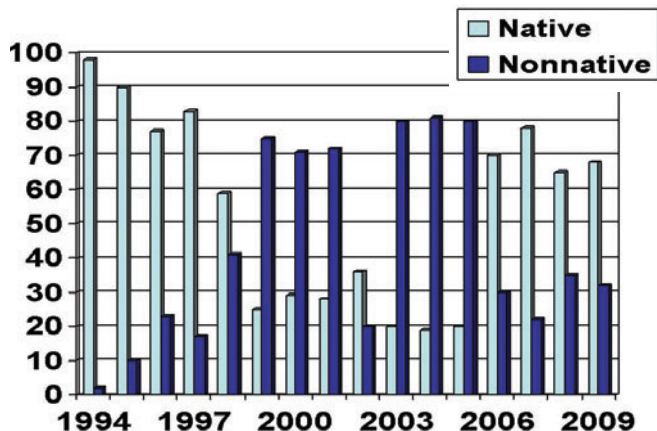


Figure 9.33—Native to nonnative percentages of fish assemblages, Burnt Ranch, UVR, 1994-2009.

and Sycamore sites (fig. 9.31). Sampling at the seven long-term sites (1994 to 2004) and at the Burnt Ranch and 638 Road sites prior to mechanical removal treatments provide, in part, a frame of reference (figs. 9.12 and 9.24A).

In general, following the 1992 to 1993 low recurrence, 75-year flood events (fig. 9.22), the proportion of native fish species in the total fish assemblage was sustained for two to three years (fig. 9.12). It should be pointed out that a smaller, five- to seven-year event occurred in March 1995 (fig. 9.22), and by 1997 the non-native component of the total fish assemblage became dominant and remained so even into spring 2005 following flooding in winter of 2004/2005. Instantaneous peak flows in these recent flood events immediately prior to instituting the modified mechanical treatment in 2006 were somewhat lower than those that occurred in winter of 1992/1993 (fig. 9.22, Rinne 2005).

Similar to the initial years of long-term monitoring from 1994 to 1997, native fishes again became dominant (80% of total assemblage) by spring 2006 with increased recruitment of the three larger-size longer-lived suckers and chubs the previous summer (fig. 9.28). However, these proportions immediately dropped considerably by the spring 2007 sampling. An elevated winter flow event in February

2008 produced peakflows from $70.8 \text{ m}^3 \text{ sec}^{-1}$ ($2,500 \text{ ft}^3 \text{ sec}^{-1}$) at the Paulden gauge (fig. 9.22) to an estimated $169.9 \text{ m}^3 \text{ sec}^{-1}$ ($6,000 \text{ ft}^3 \text{ sec}^{-1}$) at the Sycamore Canyon site. These flows stimulated increased spawning and recruitment and resulted in natives again dominating when all sites were averaged, despite reduced numbers (fig. 9.28).

In summary, it appears that peak flow events dramatically impact the relative proportions of native and nonnative fishes in the total fish assemblage (Rinne 2005). However, two of the seven long-term sites—Burnt Ranch (treatment site 7) and 638 Road (treatment site 3)—sustained mechanical treatment in 2006. These two treatment reaches and the other five combined with the smaller peakflow events in late winter 2008 have contributed to the native component being sustained in the UVR from 2008 to 2009 (figs. 9.30 and 9.31).

Spring 2009 sampling and removal of nonnative species revealed: (1) nonnatives dominated the fish assemblages in three of four sites in the 638 Road reach and conversely; (2) natives dominated the fish assemblage at all sites in the Burnt Ranch reach (figs. 9.32 and 9.33). This inverse relationship very likely results from the lack of treatment at sites in the 638 Road reach in summer 2008. In addition, there was some channel disturbance at an estimated half of site 3 because of the installation of an underground gas pipeline. Removal efforts in June 2009 demonstrated both the impact of the absence of summer 2008 removal and the positive effect of the two months prior treatment in April 2009.

Mechanical removal efforts that were re-initiated in spring/summer 2006 and continued through June 2009 that employed a modified approach addressing shortcomings of the 1999 to 2003 pilot efforts (Rinne 2001b, 2006) appear to have a positive effect on the native fishes (figs. 9.26 and 9.27). It should be pointed out that only the two native suckers and the one native chub were represented in the samples. The smaller-sized, short-lived daces—longfin (*Agosia chysogaster*), speckled (*Rhinichthys ocululus*) and threatened spikedace (*Meda fulgida*)—have not been represented in samples.

The initial removal efforts were started a year after the winter 2004/2005 flood events and all eight removal reaches revealed a predominance of natives (figs. 9.34 to 9.38). Native species dominated the fish assemblages in 2006 and 2008. However, nonnatives dominated at three to four of the eight removal sites in both 2007 and 2009. Nonnatives also dominated the fish assemblages at the control site (R-0) from 2006 to 2008. Overall, the native component in spring 2009 averaged 73% at sites where they were predominant as a result of flooding the previous year. Considering all sites, native dominance went from 58 to 89% in 2006 to 53 to 90% in 2009 (figs. 9.34 to 9.38). Nonnative fishes varied from 33 to 63% at control sites for the same period.

Total fish numbers captured during removal continued to decline as has occurred previously during drought periods at long-term monitoring sites (1997 to 2003, Rinne 2005; fig. 9.28). Total numbers fell below 1000 individuals in spring 2009, or less than 25% of the total numbers in 2006. Of the nonnative fishes, red shiner (*Cyprinella lutrensis*) continue to dominate (60%) (fig. 9.29). However, their numbers have declined to a third of those in 2006. It is obvious from figs. 9.28 and 9.29 that if red shiners were removed from the nonnative totals, the native compared to nonnative percentages presented above would change dramatically in favor of the native component.

Similar to the time period of 1994 to 2004, total fish numbers in the UVR at the eight mechanical treatment removal sites dropped 80% from 2006 to 2008 (fig. 9.28). The decline is largely reflective of the drop in native fish numbers from over 4,000 individuals captured in spring 2006 to 1,200 in 2008. By contrast,

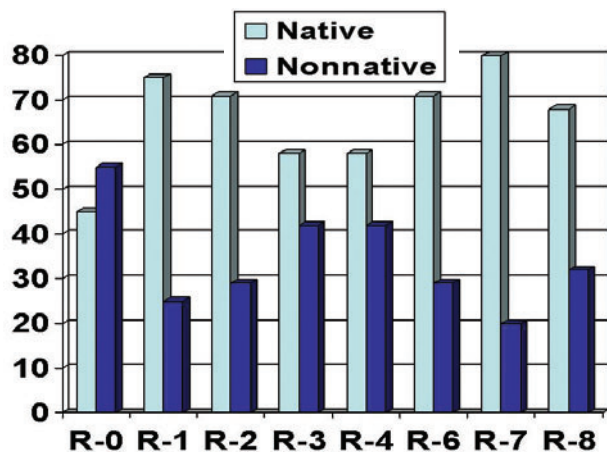


Figure 9.34—Native to nonnative percentages of fish at seven mechanical removal reaches (R-1 to R-8) and one control reach (R-0), UVR, 2006. (R-5 was not sampled).

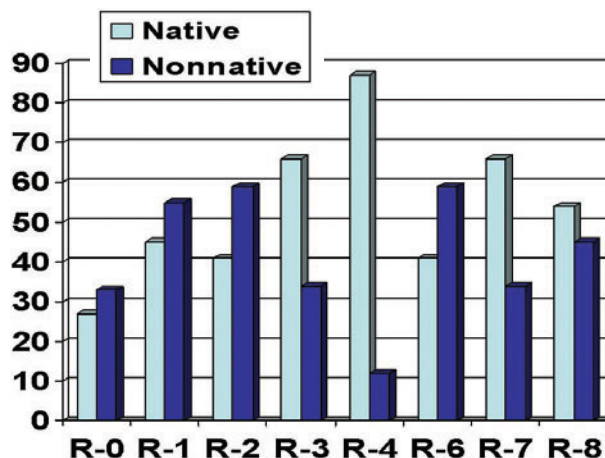


Figure 9.35—Native to nonnative percentages of fish at seven mechanical removal reaches (R-1 to R-8) and one control reach (R-0), UVR, 2007. (R-5 was not sampled).

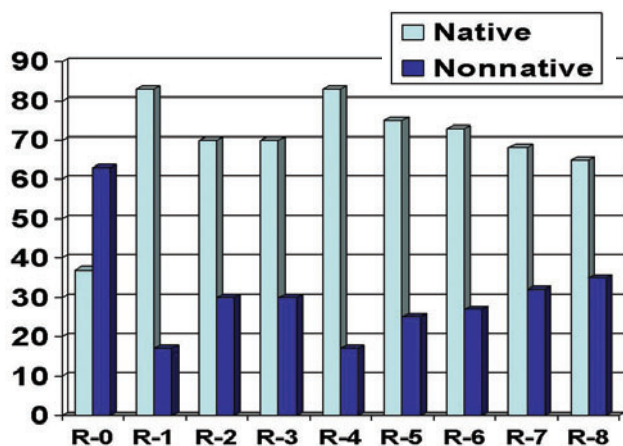


Figure 9.36—Native to nonnative percentages of fish at eight mechanical removal reaches (R-1 to R-8) and one control reach (R-0), UVR, 2008.

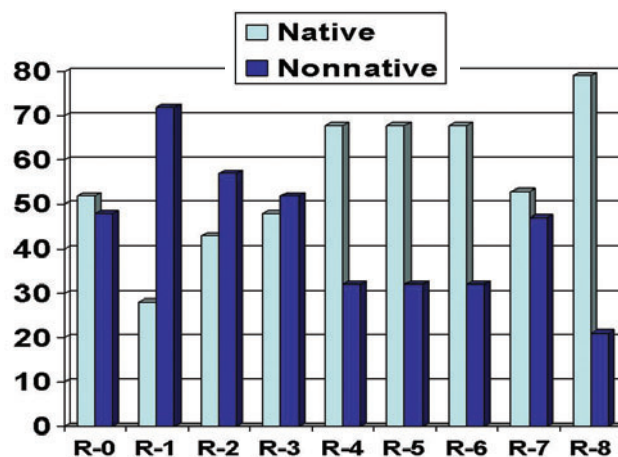


Figure 9.37—Native to nonnative percentages of fish at eight mechanical removal reaches (R-1 to R-8) and one control reach (R-0), UVR, April 2009.

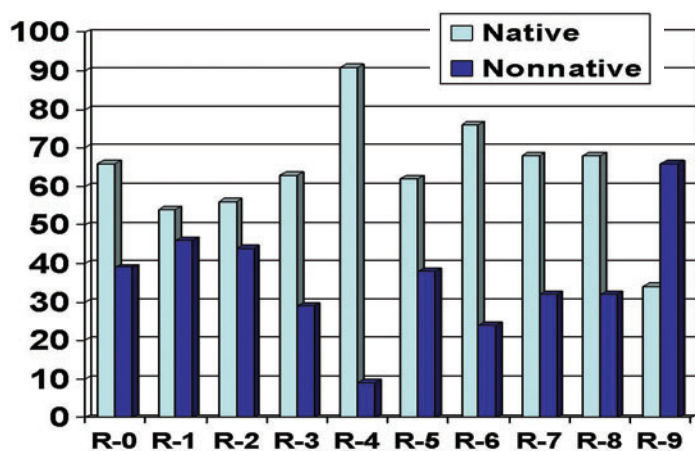


Figure 9.38—Native to nonnative percentages of fish at eight mechanical removal reaches (R-1 to R-8) and two control reaches (R-0 and R-9), UVR, June 2009.

nonnative species in removal sites were lowest (850) in 2006, increased to 1,270 in 2007 and then plummeted to around 400 in spring 2008 (fig. 9.28). Of the nonnative species, red shiner plummeted most (-75%) from 2006 to 2008, green sunfish dropped 74%, smallmouth bass 71% and yellow bullhead basically remained constant from year to year 59, 77, and 59% with no apparent response to mechanical removal (fig. 9.29).

Summary and Conclusions

In summary, native/nonnative ratios in the mechanically treated reaches are very similar to those resulting from large flow events. Instantaneous peak flows in September 2004 and again in early 2005 were 50% of those in 1992/1993 and produced similar results in ratios in native and nonnative fish assemblages. However, the drop within two years to a 50/50 ratio in spring 2007 and again in spring 2009 at the long term sites suggest that the fish assemblage was again rapidly returning to that dominated by nonnative fish species. Only with additional monitoring at the long-term sites and continual removal in all three seasons—spring (April), summer (June) and autumn (October)—and in absence of significant flooding will it be more to say that mechanical removal has a high probability of benefitting native fish sustainability, albeit only that of the larger sized, longer-lived suckers and the roundtail chub.

Data on fish assemblages over the past 15 years at long-term sites and fish assemblage response to mechanical removal and flow regimes indicate but do not confirm that mechanical removal is an effective tool to at least sustain the larger-sized, longer-lived native fishes between and in combination with instantaneous flow events. Red shiner, although numbers have been reduced three-fold since inception of treatment in 2006, appear to be the most difficult nonnative species to reduce to the levels of the other three larger-sized nonnatives (i.e., 50 to 100 in a total sample period at all eight treatment sites).

Accordingly, it is strongly recommended that both the long-term monitoring be continued for several years along with mechanical removal to determine if over the next few years. Perhaps, in absence of significant, influencing flow events, native fishes may dominate the fish assemblage again. Efforts should be made to assure that a third removal in autumn is effected at both the 638 Road and Burnt Ranch reaches. The latter has sustained only the spring treatment in 2007 and 2008 because of a lack of resources.

Hydrographs, Geomorphology, and Fish Assemblages

The current working hypothesis is: “Flooding is very important for sustaining native fish assemblages where nonnative fishes are present in Southwest rivers and streams.” Minckley and Meffe (1987) first suggested this hypothesis for Southwest streams and Stefferud and Rinne (1995), Brouder (2001), Rinne and Stefferud (1997), and Rinne (2005) corroborated and advanced this hypothesis for the UVR. Minckley and Meffe (1987) contrasted differences in flood hydrology between Southwest arid lands and those of lowland mesic regions of the central and eastern United States. They reported that most of the annual water yield in Southwest systems was produced during high discharges in brief periods of time, whereas low discharges produced a far greater proportion of total yield from mesic systems. They suggested the differential effects of these patterns on fishes were significant

to the native/nonnative mix of species now present in the Southwest. The authors concluded that native Southwest fishes were better adapted to withstand the effects of large floods than nonnative fishes. During severe floods, invasive fish species were either displaced or suffered mortality, whereas native species maintained position in or adjacent to mainstem channel habitats, persisted in micro-refuges, and rapidly re-colonized post-flood.

Building upon the Minckley and Meffe (1987) hypothesis, Rinne (2005) furthered this concept in the UVR. Rinne and Miller (2006) in a comparison of the Verde and Gila Rivers, documented that flow regimes are very basic to native fish sustainability and may have an overriding influence relative to human-induced activities, including fisheries and riparian management. Most recently, Propst and others (2008) further confirmed this hypothesis using 20 years of data at sites complementing those sampled by RMRS scientists since 1999 (Rinne and others 2005a, 2005c). The UVR is an example of a flood-dependent, disturbance-dependent ecosystem with a mixed native and nonnative fish assemblage.

In the UVR and other streams and rivers in the western United States, alteration of the natural hydrographs through natural climatic factors and human-induced impacts such as dams, diversions, pumping, and land use, and introduction of non-native species of fish appear to interact to affect stability and integrity of native fish populations (Minckley and Deacon 1991). In all but a few streams in Arizona and the Southwest, these factors have dramatically reduced or eliminated native fish species and modified fish assemblage structure and dynamics, often within a few years after the action. Streams with purely native fish assemblages, or where the native component of the assemblage remains dominant, are a rarity in the region. The mechanisms that sustain native species in the presence of nonnative species are not well understood but are thought to be related, in part, to natural flow regimes characterized by extremes of drought and base flows and peak flow events and flooding. These climatic and hydrologic conditions characterize streams in the Southwest (Minckley and Meffe 1987), including the UVR (Rinne and Stefferud 1997; Brouder 2001). How flow regimes interact with the presence of competitive/predatory nonnative fishes is not well defined and merits further research. Clarification of this question is important at this time because of the ever increasing demand for water for a growing populace and the generally imperiled status of Arizona's native fish fauna.

Although a number of studies on fishes and their habitats have been undertaken in the Verde River as a whole, not one has investigated the long-term interrelationships of the native/nonnative fish assemblages relative to flood disturbance events and to each other. The relationship of base and instantaneous peak (flood) flows to sustainability of native fishes in the UVR is a prerequisite for inputs for water management plans for the system. The headwater Big Chino Valley aquifer is a major source of baseflow for the Verde River above Sycamore Creek; and there are proposals to extract its water to supply current municipal needs and future development in the watershed. Depletion of discharge from the aquifer could potentially alter base flow in the river and will dramatically and chronically affect sustainability of native fishes (Rinne 2001a, 2001b, 2001c, 2001d). However, the unresolved question of their effects on the native fish assemblage has deferred approval by resource agencies. Continued growth in the Verde River watershed will probably increase pressure for water development and diversion from the UVR. Resource management agencies must have reliable data in time and space to chart a course of action that will sustain native fishes in the UVR.

Relationships of Hydrographs and Fish in the UVR

Data from previous studies in the UVR indicated that following almost a decade (1984 to 1992) of drought and low flows (fig. 9.13), nonnatives only comprised 12% of the total fish assemblage (Marty Jackle, USDI Bureau of Reclamation, pers. comm.; USDI Fish and Wildlife Service 1989). Fishes were not sampled after the 1992/1993 flood events until spring 1994. Notwithstanding, following the massive flooding in winter 1992/1993, nonnatives were yet reduced in 1994 to only 4% of the total fish community. Therefore, Minckley and Meffe's (1987) hypothesis on the mechanism of interactions and control of the native and nonnative fish assemblages in the Southwest was partially correct. Not only the nonnative fishes are impacted by flooding, but the native fish are affected as well. However, immediately post-flood, native fish rebound rapidly; nonnatives also increase, but more slowly, and some ratio of the two is established. The ratio appears, in part, to be dependent on the subsequent annual hydrographs and relative native/nonnative fish response. For the first three years post-flood this ratio was consistently 85:15 (fig. 9.13), native to nonnative species. However, the fact that nonnative fish comprised only 12% of the total fish community in the late 1980s (see the previous section) after a four-year period of low flow suggests other mechanisms may influence the native/nonnative ratio in the UVR.

Based on the data from the Verde River following 75- (1992/1993) and 7-year (1995) flood events (fig. 9.13), it appears that floods of these magnitudes negatively impact and dramatically reduce both native and nonnative fish communities. The seven-year event that occurred in early spring (March) 1995 reduced the native fishes in the UVR by 86% and the nonnative community by 89% (tables 9.1 and 9.2). However, the native/nonnative fish assemblage composition ratio remained almost identical between years, respectively (82:18 and 85:15). Further, by spring 1996, the native fish component had increased, on average, by over 700%. In contrast, invasive fishes, despite marked increases in red shiner and smallmouth bass, increased by an average of only 71%.

Green sunfish was the only nonnative species to increase in absolute numbers in the UVR fish assemblage between 1994 and 1995 (5 to 29 individuals) (Rinne and Miller 2006), suggesting that it was a result of a differential magnitude of flooding over the entire study area. The flood in 1995, in contrast to that in 1993, affected mostly the reach of river downstream from Sycamore Creek that was not recorded by the Paulden gauge. The majority (26 of 29 individuals) of the increase in green sunfish numbers occurred in the upper reaches of the river (Sites 2 through 5). The peak discharge at these sites, as indicated by the Paulden gauge, suggests that flows were likely below the threshold to negatively impact this species. Similarly, the input of fathead minnow that originated in watershed stock tanks as a result of increased precipitation and runoff in spring 1994 may have been the reason for the appearance of this species immediately post-flood in the spring 1993 sample at Burnt Ranch (Sponholtz and others 1998). By autumn 1993, fatheads had disappeared and the native fish component reached 96% of the fish community.

Precipitation and stream hydrographs are stochastic and unpredictable in the Southwest and floods of significance (i.e., $>400 \text{ m}^3 \text{ s}^{-1}$ or $14,125.6 \text{ ft}^3 \text{ s}^{-1}$) in the UVR appear to occur randomly. Cycles of flood and drought, however, are evident, and ensuing years of low flows were more probable following the 1993 and 1995 floods. Further, based on historic hydrographs, the probability for a period of low flows also increased after 1995.

Typical of desert streams and rivers, flooding can be significant in the Verde River. Larger floods, estimated to have recurrence intervals of 50 to 60 years (USDI Geological Survey 1992), appear to “reset” the channel morphology and aquatic habitats by eroding stream banks and restructuring and invigorating substrate materials (Rinne and Stefferud 1997; Rinne and Miller 2008). After each flood, total abundance of fish was reduced and the population structure of the various species was altered. Based on data collected after winter 1993 and 1995 flood events (both 75-year and 7-year return intervals), the floods reduced both native and nonnative species. Nevertheless, natives yet comprised 85% or more of the fish assemblage. Following these events, the natives rebounded quickly in numbers in response to the restructuring or invigorating of substrate materials (Mueller 1984; Rinne and Stefferud 1997; Rinne and Miller 2008). Spawning success and recruitment contribute to the quick re-establishment of native fish populations’ biological and geo-morphological reference baseline. That is, riparian vegetation succession is set back to a base level, channel morphology is modified, and stream-bed materials are sorted and re-arranged.

Although spawning success and recruitment after floods probably contribute to the quick re-establishment of native fish populations, nonnative species begin to increase at the same time (table 9.2). Nonnative fish lowered numbers also would favor a rapid increase in native fish numbers because of the lack of competition and more probably predation (Minckley 1983; Rinne 1995b; Rinne and Alexander 1995). Further, recovery rates of nonnative fish populations were variable. Red shiner was markedly reduced by the seven-year flood event, common carp were less reduced, and smallmouth bass sustained their numbers.

As previously discussed, long-term (>five years) studies of native fish populations in low elevation desert rivers of similar or larger size are non-existent. Several long-term studies in smaller streams are ongoing: Aravaipa Creek, Graham and Pinal Counties, Arizona, is the subject of a 25-year record of fish community dynamics (W. L. Minckley, pers. comm.), and there are seven-year records of native fishes and habitat associations for several streams in the upper San Francisco and Gila River drainages in New Mexico (D. L. Propst pers. comm.; Rinne and others 2005b). Whereas these existing datasets may be useful for comparing effects of floods and droughts on native species, none of the studies are in a stream reach that sustains a significant nonnative fish component. Thus, the Verde River research and monitoring effort is unique in the Southwest and offers an excellent opportunity to test and refine current hypotheses relative to native/nonnative fish interactions, as influenced by drought and flooding over the long term.

Based on the data in the UVR (Rinne 2005) and from the upper Gila (Rinne and Miller 2006; Propst and others 2008), the native fishes of Southwest desert rivers and streams appear to be disturbance species. Disturbances can come in both natural and anthropogenic forms. Disturbance is defined here as a “specific, significant, and variable spatial/temporal event that dramatically alters structure and function of both physical and biological components of an ecosystem.” Periodic instantaneous peak flow events in Southwest aquatic ecosystems certainly qualify as natural disturbance events. Further, as suggested above, the native fishes very likely can be considered “disturbance species” that directly and markedly are influenced by such periodic disturbance events to sustain populations through time. Too often, disturbance is considered to be anthropogenic or human-induced rather than natural. Major dams, diversions, and groundwater mining, for example, qualify as significant sustained disturbance events (Rinne and Minckley 1991). However, other less marked impacts imposed by intrinsic and extrinsic human-induced uses and impacts such as riparian/stream corridor and landscape or watershed uses are likely given more significance than they merit. Rinne and Miller (2008) suggested

that overriding, natural hydrologic/geomorphic influences are likely more important in terms of creating habitat essential for sustaining fishes assemblages than is simple increase in riparian vegetation that may result from riparian restoration activities such as livestock grazing removal.

Verde River Hydrographs

A study of fish assemblages of the UVR relative to abiotic and biotic factors was conducted in 1997/1998 by RMRS personnel with Arizona Heritage Foundation funding (Rinne and others 1998). The primary objective of the research was to determine the relative influence of the hydrograph and introduced fishes in delimiting the relative native to nonnative composition of the total fish community. The results of the three years of study (Rinne and Stefferud 1997) showed that native fishes respond positively to flooding. Further, Rinne and others (1998) reported that the hydrograph and the lack of flooding over the entire Verde River have enhanced nonnative over native fishes (figs. 9.12 and 9.17). By spring 1998, after three years (1996 to 1998) of no significant (i.e., greater than bankfull) flow events, the relative abundance of native fishes (>70%) was reduced to the extent that nonnative fishes comprised this same relative percentage of the total fish community. Rinne and others (1998) concluded that flooding was essential for sustaining a native fish fauna in a river system such as the UVR with nonnative fishes present.

The threatened spinedace populations paralleled the overall population trend of native fishes in the UVR (Rinne and others 1998). The species was common in samples at four of the seven established sampling sites in the spring of 1994, rare by the spring of 1996, and absent in the spring of 1997 (fig. 9.15A). Therefore, flow regimes appear to be important to essential for sustaining not only native fishes in general but specifically populations of spinedace, especially in the presence of nonnative fishes.

Rinne and Stefferud (1997) concluded after three years of sampling that the river hydrograph was more influential than nonnative fishes for sustaining a native fish component in the UVR. Notwithstanding, it appears that the interaction of the two factors combine to legislate the relative native to nonnative composition of the overall fish community. The lack of any significant flooding and sustained base flow since 1996 resulted in nonnative fishes increasing significantly to comprise the majority (>70%) of the fish community (figs. 9.12 and 9.17). Accordingly, in the absence of significant flow events, nonnative fish species quickly and markedly became dominant. Only with the advent of flooding of at least the 1995 level (five+ year recurrence) can this hypothesized relationship of hydrograph and nonnative fish abundance be more completely defined. This relationship needs to be further research in the upper Verde and Gila rivers.

In summary, if RMRS studies had been conducted only during 1994 to 1996, flooding could be readily defended as the controlling factor in delimiting spinedace populations in the UVR. By contrast, studies during only 1997 to 1999 could lead to the conclusion that nonnative fishes are the controlling factor in delimiting spinedace numbers in the UVR. These data emphasize the importance of continual monitoring up to and following the next flood event (five-year recurrence or greater).

Aquatic Macrohabitats and Native Fishes

Methods

Analysis and description of aquatic macrohabitats in the UVR were delineated by Rinne and Stefferud (1996) and Rinne (2003b). These habitats were sampled for fishes in sequence, progressing in an upstream direction. Standard sampling methods were used including backpack DC electro fishing units, seines and dip and nets to sample macrohabitats. All fish captured within a macro-habitat were identified to species and counted. Total length of all specimens greater than 150 mm (about 6 in) was obtained. A sub-sample of individuals greater than 150 mm (6 in) were measured and weighed to define population demographics. Once measured, all fish except for periodic voucher specimens of nonnatives were returned alive to the stream.

Initially, each macrohabitat was numbered in sequence and subjectively classified based on ongoing fish-habitat studies in the upper Gila River, New Mexico, as lateral scour, mid-channel and backwater pools (POOL); glides (GLDs); runs (RUNs); glide-runs (GRUNs); low gradient riffles (LGRs); high gradient riffles (HGRs); and edge waters (Rinne and Stefferud 1995; Sponholtz and Rinne 1997). Length, width, depth, velocity, and substrate parameters for each macrohabitat were estimated and recorded. Gradients were measured with a laser level, substrates were measured using methodology outlined in Bevenger and King (1995), and velocity was measured with a direct readout digital current meter.

Sampling was designed to expeditiously gather data that would describe fish assemblages and characterize the habitat. With these data, it was possible to initially delineate relative, quantitative macrohabitats (mean length, width, depth, velocity, and substrate). However, refining the definition of aquatic macrohabitats was considered necessary. Accordingly, increased sampling intensity and rigorous statistical analyses resulted in greater potential accuracy in classifying aquatic macrohabitats in the UVR (Sponholtz and Rinne 1997). Further refinement of fish-habitat relationships needs to be researched at established study sites. Changes in habitat at study sites can conceivably be defined and then compared to fish assemblage dynamics.

Initially, aquatic habitats were visually and subjectively defined for sampling purposes (Stefferud and Rinne 1995). Aquatic macrohabitats in the UVR were more objectively defined based on physical measurement of gradient, velocity, depth, and substrate composition by Rinne and Stefferud (1996). Measurements were then statistically summarized by previous subjective classifications. Gradients successively paralleled subjective, lotic habitat classifications: glides $\leq 0.3\%$, runs ≥ 0.3 to 5% , low-gradient riffles 0.6 to 1% , and high-gradient riffles $\geq 1\%$ (Rinne and Stefferud 1996). Velocities in pools displayed a high degree of variability (0 to 70 cm s^{-1} or 0 to 27.6 in s^{-1}), and velocity averaged the same as glide habitats suggesting shortcomings in subjective classifications. Pool classification difficulties are attributable to the many (>12) subjective types of pools (side channel, mid-channel, and scour pools; Stefferud and Rinne 1995) that were grouped within the pool category. Mean velocity increased as one moved up the lotic scale from glides and runs up through high-gradient riffle habitats (Rinne and Stefferud 1996; table 9.4, fig. 9.39). Similarly, substrate type changed from a greater percentage of small material (sand-gravel) in glides and runs in comparison to that in low- and high-gradient riffles (Rinne and Stefferud 1996).

Table 9.4—Substrate composition of macrohabitats as determined by pebble counts in the Gila (1999) and Verde (1996) rivers.

Habitat	Fines	Gravel	Pebble	Cobble	Boulder
	%	%	%	%	%
Gila River					
POOL	37	32	20	9	2
GRUN	27	51	15	6	1
LGR	10	48	28	12	2
HGR	6	35	37	20	2
Verde River					
GLIDE	30	55	10	5	0
RUN	25	30	25	20	0
LGR	20	30	25	25	0
HGR	0	15	30	50	5

Increased macrohabitat sampling was established in 1996 sampling by placing sample transects every 3 m (9.8 ft) within a habitat type (Sponholtz and Rinne 1997). Previously (1994/1995), habitat parameters were measured at only three transects/aquatic macro-habitat, irrespective of length of habitat type. Normally, four to six transects resulted from this more detailed 1996 sampling. As a result, data points for depth and velocity increased three-fold in terms of frequency. In addition, laser-generated gradients were measured at all habitats after initial results were recorded at only 14 random habitats in 1995 (Rinne and Stefferud 1996; fig. 9.7). Qualitative habitat descriptions in spring 1996 were used to delineate habitat type based on selected descriptors (Sponholtz and Rinne 1997). Pebble counts (Bevenger and King 1995) were performed in all habitats, with 30 “hits” used to estimate substrate type. Rigorous statistical analyses employing analysis of variance (ANOVA) and Classification and Regression Tree (CART) were performed on the data.

The ANOVA indicated higher variability in estimates of habitats among transects than among individual points; therefore, data were averaged for transects prior to statistics being calculated. Results of analysis suggested that depth and velocity were not sensitive enough to detect differences between either the high- and low-gradient riffle types or between run and glide habitats. Gradient was the only variable that most consistently (71% of the time) and clearly defined aquatic macro-habitat type. Further, when used with depth and velocity, gradient became an even more powerful tool, providing greater than 80% correct classification of aquatic macro-habitat types.

Aquatic Macrohabitat Comparisons in the Gila and Verde Rivers

The same fish and habitat sampling protocols were initiated on the upper Gila River in southwestern New Mexico. These waters have basically the same fish assemblage as that of the UVR. Therefore, comparisons of fish assemblage and habitat changes can be made.

Estimated velocities of 187 aquatic macrohabitats in the upper Gila River are shown in fig. 9.39. The mean velocity in POOLs was 20 cm s⁻¹ (7.9 in s⁻¹) compared to 67 cm s⁻¹ (26.4 in s⁻¹) for HGRs. GRUNs and LGRs displayed intermediate mean velocities of 38 to 52 cm s⁻¹ (15 to 20.5 in s⁻¹). The mean depths of these same habitats were inverse to velocities, ranging from about 60 cm (23.6 in) for POOLs to 20 cm (7.9 in) for HGRs.

Gradients for these same habitats in the upper Gila were least for POOLs (mean = 0.3%), were greatest for HGRs (mean 2.22%; fig. 9.40), and were parallel to

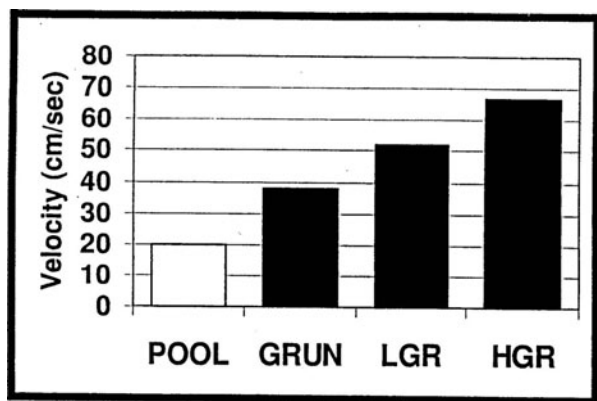


Figure 9.39—Streamflow velocities in cm sec^{-1} of habitats, Upper Gila River.

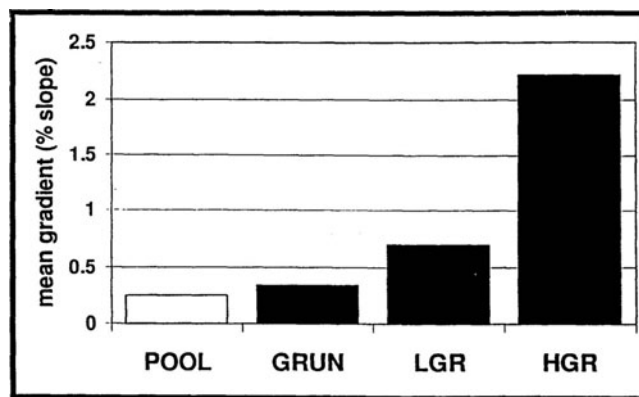


Figure 9.40—Slopes of POOL, GRUN, LGR, and HGR habitats in the Upper Gila River.

velocity and inverse to depth. Because gradients were suggested by Sponholtz and Rinne (1997) to be the best descriptors of aquatic habitat types in the UVR, Arizona, these data were analyzed for the upper Gila (fig. 9.40). Gradients of 49 HGR habitats ranged from 0.75 to 7% slope and averaged 2.22%. About 35% of all HGRs were greater than 2% slope, 18% were greater than 3% slope, and more than half (57%) were 1 to 2% slope. These values were very similar to gradients delineated for HGR in the UVR (Rinne and Stefferud 1996). Half of the HGR gradients in the UVR were between 1 and 2%, and 25% fell between 2 and 3% slope. By comparison, attempts at refinement of aquatic habitat definition in the UVR by Sponholtz and Rinne (1997) indicated that HGRs ranged from 0.56 to 4.39% and averaged 1.83% or near the mean (2.22%) for the upper Gila River. Accordingly, comparison of macrohabitat data between the Verde and Gila rivers is valid.

The 52 LGR samples ranged from 0.05 to 4% slope, averaging 0.70% (fig. 9.40). Only 20% of all estimated values were greater than 1% slope, and half were between 0.5 and 1%.

Forty percent (21) fell between 0.5 and 0.75% slope, and 33% were less than 0.5% slope.

Rinne and Stefferud (1996) reported that all estimated gradients for GRUNs (glide/run) in the UVR also were between 0.5 and 1% slope. Sponholtz and Rinne (1997) reported that LGRs in the UVR ranged from 0.72 to 2.38% slope and averaged 1.33% slope.

GRUN habitats ranged from 0.05 to 2.8% and averaged 0.35% slope in the upper Gila River (fig. 9.40). Only seven (12%) estimated slopes were greater than 1%. Thirty-eight percent of the estimated slope values for GRUNs were less than 0.25%; 48% (28) ranged between 0.025 and 0.5% slope. Rinne and Stefferud (1996) listed glide and run habitats separately, however one-third of their estimates fell between 0.25 and 0.5%. By comparison, Sponholtz and Rinne (1997) listed run habitats averaging 0.7% and glides 0.5%, or somewhat higher than the estimates from the upper Gila.

Finally, 28 pool habitats ranged from 0.0 to 0.2%, one-third was below 0.1%, and one-third was between these two values (fig. 9.40). Rinne and Stefferud (1996) listed no values for pools, but Sponholtz and Rinne (1997) listed pools ranging from 0.0 to 2.05% slope and averaging at 0.9% somewhat higher than in the upper Gila. Rinne and Stefferud (1996) and Sponholtz and Rinne (1997) found velocities of 22 and 18 cm s^{-1} (8.7 and 7.1 in s^{-1}), respectively. These data indicate and reflect that many pools in both the upper Gila and Verde commonly have slopes greater than zero and have positive current velocities.

Table 9.5—A comparison of percent availability of substrate category in the Gila and Verde Rivers and that used by spikedace.

Substrate	Spikedace		Total substrates	
	Present	Absent	Gila River	Verde River
	%	%	%	%
Sand	12	5	20	18
Gravel	39	19	43	40
Pebble	19	28	25	2

The HGRs consisted of gravel to cobble materials in both rivers. LGR habitats contained primarily pebble-gravel substrates in the Gila compared to gravel to cobble materials in the Verde. GRUNs in the Gila were almost 80% sand and gravel compared to sand- cobble in the UVR. Pools had sand-gravel substrates. Comparative substrates in the upper Gila and Verde for the respective habitat types are shown in table 9.5).

Rinne and Stefferud (1996) reported that HRGs in the UVR had a mean of about 50% cobble-boulder, 30% pebble, 15% gravel, and 0% sand. LGRs were about 25% cobble-boulder, 25% pebble, 30% gravel, and 25% sand. Finally, glides contained basically none of the cobble-boulder category, 10% pebble, 55% gravel, and 30% sand.

Macrohabitat Utilization by Native Species

Overall, native fishes were collected in pool habitats in the UVR (Rinne and Stefferud 1996). However, this association can be attributed, in part, to the predominance of two of the three large species (Sonora sucker and roundtail chub) which comprised 30 to 50% of the native fauna in 1994 and 1995, respectively (Rinne and Stefferud 1996) and were captured 11 and 61% of the time. Further, desert sucker (which comprised 25 to 30% of the native fauna) were captured in pools 20% of the time (Rinne and Stefferud 1996; fig. 9.41). Finally, the great variability in velocity of pool habitats that were comprised of multiple pool categories increased the probability that a greater number of individual fishes would be captured in pool habitat types. Longfin dace were captured primarily in low-gradient riffles and runs (fig. 9.42), spikedace in runs and glides (fig. 9.43; Neary and others 1996), and speckled dace in low- and high-gradient riffles (fig. 9.44).

Macrohabitat Utilization by Nonnative Species

Nonnative species primarily (29%) used glide habitat. Red shiner were collected mainly in GRUN habitat. Green sunfish, yellow bullhead, and smallmouth bass were taken in 91 habitats—comprised of 45% pools, 22% glides, 20% runs, and 13% riffles. Common carp was taken in pools over half the time. In addition, Rinne and Neary (1997) demonstrated that yellow bullhead and smallmouth bass frequently occupied streambank habitat that provided cover. Further, where these nonnatives were present, native species were reduced or absent. These studies are ongoing but suggest possible competition for habitat and/or predation effects of these two nonnative predatory species on the smaller-sized native species such as longfin dace and spikedace and YOY and juveniles of the desert and Sonora suckers and roundtail chub. Unpublished laboratory data using bullhead and smallmouth bass as predators and red shiners as a surrogate for both the small cyprinid

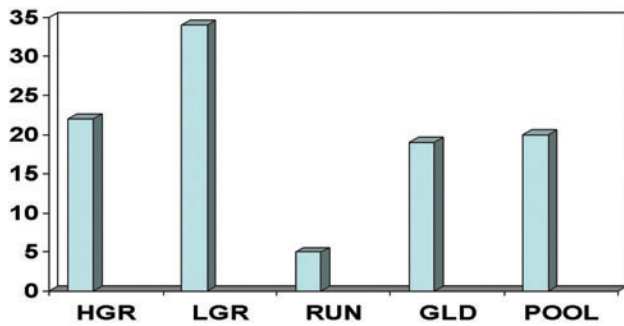


Figure 9.41—Macrohabitat utilization by desert sucker, UVR; number = 2,717.

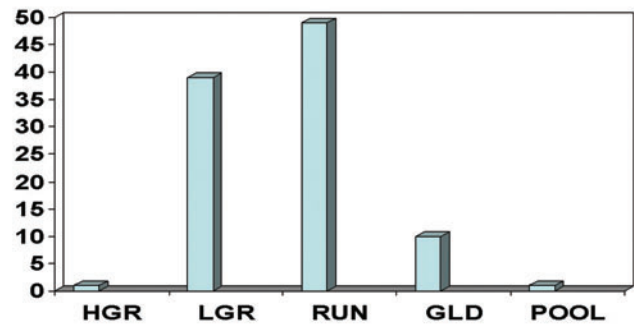


Figure 9.42—Macrohabitat utilization by longfin dace, UVR; number = 1,814.

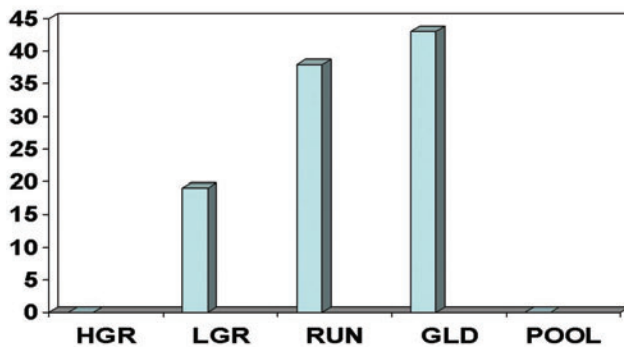


Figure 9.43—Macrohabitat utilization by spikedace, UVR; number = 1,177.

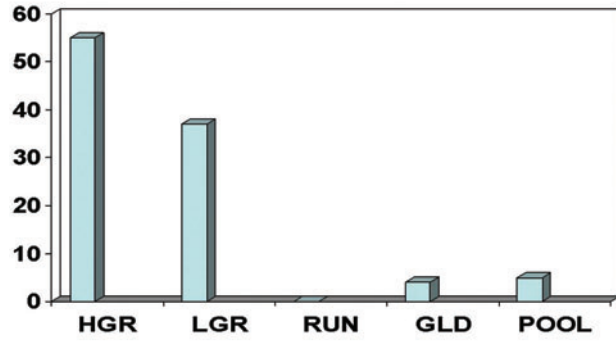


Figure 9.44—Macrohabitat utilization by speckled dace, UVR; number = 117.

species and YOY of the three large cyprinids suggest the predation effect on native species potentially could be substantial.

Stream Hydrology and Fish Habitat

Based on annual peak discharges and mean annual discharges over the past three decades, flows in the UVR, typical of Southwest rivers and streams, were highly variable (Stefferd and Rinne 1995; Rinne and Stefferud 1996). Neary and Rinne (1998) documented that base flow in the UVR has increased over the past two decades. Cycles of flood and drought appear to follow a 7 to 10 year periodicity. Although there are no specific data on channel morphology prior to the 1993 flood, morphology was observed to change dramatically after the 75-year event. The seven-year event in 1995 also altered channel morphology but to a lesser extent as evident in changes in the relative abundance of habitat types (see Chapter 5). Based on aquatic macrohabitat type, most of the major sample reaches change from year to year. A lack of significant flood events following 1995 sampling and ensuing (1996 to 2004) low, drought flows appear to have significant effects on macrohabitats (Sponholtz and Rinne 1997). Stream channels have narrowed and aquatic macrophytes have encroached from stream margins, reducing mean stream width. Depths of many macrohabitats also have been reduced because of a lack of scouring (Rinne and Miller 2008). The relationship of stream channels and flow events need to be further analyzed with data from the UVR and the very large dataset from the upper Gila River.

Status and Habitat of Spikedace

The threatened spikedace (*Meda fulgida*) was once widespread and locally abundant in streams and rivers of the Gila River Basin (fig. 9.4). It is a Federally-listed threatened species, occurring in four isolated stream and river systems in Arizona and New Mexico. Based on 15 years of study of the Verde River population, spikedace declined dramatically in abundance, even to the point of non-detection in samples since 1997 (Rinne 1999a). In 1999, calculated probabilities of extinction (0.8 to 0.9) suggest that the species is extirpated from the UVR. Based on current data, river hydrograph, habitat changes, and nonnative fish play interactive roles in this decline. Complicating determination of relationships and causal factors has been the removal of livestock grazing (see the “Grazing Management Changes” section) from the Verde River riparian stream corridor and the resulting changes in riparian stream habitat.

From an intra-stream perspective, monitoring for spikedace on the UVR should be continued and intensified. In the event of reappearance, local experimental suppression or reduction of nonnative fishes in these reaches should be considered to observe both the spikedace and the native fish assemblage response. Also, refugia streams should be identified and evaluated, culture techniques should be developed, and research priorities should be established. Research should be continued and should include comparative studies with the upper Gila River, to refine the level of understanding of the relationships and interactions of flow regimes, native/nonnative fish abundance, and land use activities.

Description and Biology

The spikedace is a slender, laterally compressed cyprinid. The dorsal surface is typically olive gray to brown, often mottled with silvery sides and the ventral region is whitened. Males become golden on their dorsal and lateral surfaces during breeding. The basic life history of the species has been studied primarily in one stream in Arizona—Aravaipa Creek (Barber and Minckley 1966, 1983; Barber and others 1970; Turner and Tafaelli 1983; Rinne and Kroeger 1988). In addition, Anderson (1978) and Propst and Bestgen (1986) have studied the biology of the species in the Gila-Cliff reach of the mainstream Gila River in New Mexico. However, most of the information collected on the species has been from survey and monitoring activities to assess the distribution and abundance of the species in time and space (LaBounty and Minckley 1972; Anderson and Turner 1977; Barrett and others 1985) and to provide limited information on the biology of the species.

Spikedace spawn in the spring and early summer (April to June) and are principally dependent upon streamflow and attendant water temperature, normally varying in time and space. In early March, 1999, extreme coloration of males and gravid females were noted in the Gila-Cliff valley of New Mexico (personal field observations). Eggs are expelled in the water column and are adhesive when fertilized, adhering to the substrates where spawning occurs. Sand to gravel-pebble substrates have been reported where reproductively ready individuals (based on coloration and gravidity) have been collected in the UVR (Neary and others 1996), Aravaipa Creek (Barber and others 1970), and the upper Gila River, New Mexico (Propst and Bestgen 1986). Spawning habitat was described by Neary and others (1996).

Historic Distribution and Current Status

The spikedace, is a diminutive, short-lived, stream-dwelling minnow endemic to the Gila River Basin of Arizona and New Mexico (Miller and Hubbs 1960; Minckley 1973). Although once widespread in the Gila River Basin (Rhode 1980; Propst and Bestgen 1986), the spikedace is now a Federally listed threatened species (USDI Fish and Wildlife Service 1986) and occurs only at intermediate elevations (1,070 to 1,830 m, or about 3,500 to 6,000 ft) of the upper Gila River in southwestern New Mexico (Propst and Bestgen 1986; USDI Fish and Wildlife Service 1990), Aravaipa Creek (Barber and Minckley 1966; Bettasco and others 1995), Eagle Creek (W. L. Minckley, unpublished data), and the UVR (Stefferdud and Rinne 1995).

Historically, the spikedace was probably widespread and locally abundant in the Gila River Basin from low- to mid-elevation reaches of the San Pedro (type locality; Miller and Hubbs 1960), Salt, Verde, Agua Fria, San Francisco, and Gila rivers. Although present at upper elevations (1,525 to 1,830 m, or about 5,000 to 6,000 ft) of these same mainstream rivers, population numbers were likely lower. Gaps in quantified temporal-spatial information such as museum collections do not permit unequivocal assessment and delineation of historic distribution. Generally, the species was common throughout the basin and probably was locally abundant in preferred habitats. As with many western native cyprinids, fluctuations in both range and numbers of spikedace in response to regional environmental and climatic conditions are the norm (Minckley 1973). Another related, rare native cyprinid, the Little Colorado spinedace (*Lepidomeda vitatta*), ranges from abundant to rare in habitats of its native range and is extirpated from certain local stream systems (Minckley and Carufel 1967). For both species, extrinsic and intrinsic factors may likely be delimiting variations in range and abundance; however, specific factors have not been adequately defined. Further research is needed on the factors critical for sustaining this rare, threatened native fish species relative to intra- and interactive natural and human-induced factors in the Verde and Gila Rivers.

Verde River Spikedace—Recent Distribution and Abundance

As discussed previously, the UVR is a free-flowing stream with low base flows ($0.57 \text{ m}^3 \text{ s}^{-1}$ or $20 \text{ ft}^3 \text{ s}^{-1}$), only one perennial tributary (Granite Creek), and periodic extreme flood events (Stefferdud and Rinne 1995). Although adapted to the vagaries of Southwest stream dynamics, over the past six years, spikedace populations have appeared to be affected dramatically by natural variations in the Verde River hydrograph (i.e., alternating floods and drought; Stefferdud and Rinne 1995; Rinne and Stefferdud 1997). In addition, introduced nonnative fishes such as the diminutive red shiner (Rinne 1991b; Douglas and others 1994; Carpenter and Mueller 2008) and larger predatory species such as small mouth bass, green sunfish, and yellow bullhead appear to have negatively affected the species (Rinne 1991a, 1991b; Stefferdud and Rinne 1995) through their proportional increase in the total fish community from 1997 to 2008.

In addition, it has been suggested that livestock grazing can negatively impact not only spikedace habitats and abundance, but native fishes in general (Rinne 1999b, 2000). Notwithstanding, no reports or published accounts are available that document the nature and extent of this impact in the UVR or elsewhere in Southwest riparian stream systems (Rinne and Miller 2008). The only information implicating this land use activity as negatively affecting spikedace habitat and populations was a draft biological opinion (administratively withdrawn) on one grazing allotment on the UVR. In general, information is lacking on the relationships between native

Southwest cypriniform fishes and livestock grazing. This relationship is in dire need of additional, well-designed, objective, and replicated research (Rinne 1999b).

Although historically present in the Verde River, the spinedace was not re-discovered in samples until the early 1970s from the mouth of Sycamore Creek (Anonymous 1974). Large numbers of spinedace were collected by the USDI Bureau of Reclamation in the mid-1980s in the UVR while examining instream flow needs of native fishes (Barrett and others 1985). Spinedace were common in samples comprising up to 11% of the native fish fauna.

As previously stated, RMRS in Flagstaff, Arizona, commenced study of fish populations in the UVR in 1994 following major flooding (75-year recurrence event) in the winter of 1992/1993 (Stefferdud and Rinne 1995). Spinedace were most abundant in the most upstream reaches of the UVR in spring 1994 (table 9.1; Stefferud and Rinne 1995; Rinne and Stefferud 1997). Fewer were collected downstream at the Perkinsville and Black Bridge sites and at the mouth of Sycamore Creek. A reduced level of flooding (seven-year recurrence event) occurred again in March 1995. Spinedace populations pulsed once more in 1996 (Rinne and Stefferud 1997), only to decline in subsequent samplings and became absent from samples at the seven established sites by spring 1997 (Rinne 1998). Additional sampling along the entire course of the UVR in 1997 and 1998 also indicated that the spinedace population had declined markedly. In the spring of 1999, sampling in the most upstream reaches of the Verde (Bear Siding upstream) again indicated no spinedace presence. Based on these data, the species presently is very rare in the UVR. Spinedace has not been collected despite extensive efforts by RMRS and Arizona Game and Fish Department efforts for a decade (Rinne 2001a, 2001c, 2005).

Grogan and Boreman's (1998) approach was applied to assess and address more specifically the rarity and probability of extirpation of spinedace from the UVR. This methodology uses years and last year of collection versus total years of surveys to arrive at a probability that a species, based on historic collection, is extirpated. A probability (P) of 0.873 that spinedace were gone from the UVR was calculated using U.S. Forest Service data collected since 1994 and a last date of collection of 1996. A more robust data set from Arizona State University that commences with collections in 1980 and has a last date of collection of 1997 resulted in a lower but similar probability ($P = 0.832$) of extirpation of this species from the UVR above Sycamore Creek. Employing that methodology with the continued absence of the species between 1997 and 2008 results in a $P = 0.93$, or 93 chances out of a 100 that the species has been lost from the UVR.

Bahm and Robinson (2008) reported on a recent Arizona Game and Fish Department spinedace survey in the UVR from Burnt Ranch to Perkinsville (fig. 9.45). Fish biologists did not capture any individuals, making this the ninth consecutive year that spinedace have not been detected during sampling by Arizona Game and Fish Department. However, the Department survey was a general fisheries surveys (electrofishing) and did not specifically target spinedace. Although they could not conclude that the species is extirpated in the UVR, Bahm and Robinson (2008) recommended that another survey should be done specifically for spinedace in 2009 to 2010 and beyond.

Comparison of Spinedace Habitat—Verde and Gila Rivers

Spinedace occupy lentic habitats of varying depths (<1 m or <3.3 ft) over gravel and pebble substrates. The species is often found in greater abundance in shear zones where two riffle areas converge to form eddying currents (Rinne 1985b, 1991a, 1991b, 1992). In larger rivers, the species is most common in



Figure 9.45—Arizona Game and Fish Department 2008 fish survey in the UVR from Burnt Ranch to Perkinsville. (Photo by Arizona Game and Fish Department.)

GRUN habitat of moderate velocities (30 to 50 cm s⁻¹ or 11.8 to 19.7 in s⁻¹) and gradient (0.5 to 1.0%; Neary and others 1996) over gravel-pebble substrates. Boulder substrates may infrequently provide alteration or reduction of current velocity and create eddying currents that appear to attract the species. Habitat associations may vary both in time and space as well as ontogenically (Anderson 1978; Rinne 1985b, 1991b; Propst and Bestgen 1986; Rinne and Kroeger 1988).

More extensive analyses of spinedace habitat occupation have been completed in the upper Gila River (Rinne and Deason 2000). The overall objective of almost a decade of study was to delineate factors that strongly influence fish assemblage structure and through monitoring and research compare with studies conducted in the UVR, Arizona (Rinne and Stefferud 1997; Rinne 1999b; Rinne and others 2005c). In spring and early summer (March to July) of 1999, fish community structure was estimated at 18 sites (Rinne and others 2005c) from the headwaters of the Gila River, New Mexico, into Arizona near the Arizona and New Mexico border. The primary objective of the study effort in the upper Gila was to establish long-term monitoring sites. A secondary objective was to compare results of sampling in both rivers in 1999 and other years, and also to contrast and compare fish assemblages in the two river systems based on temporal and spatial changes in the native and nonnative components, especially the abundance of two threatened species. Of special interest was the abundance and distribution of spinedace and loach minnow (*Rhinichthys [Tiaroga] cobitis*) (Vives and Minckley 1990; Propst and Bestgen 1991). Because of persistence of the species in the upper Gila, these studies provide a more robust database of habitat use by spinedace than that provided by studies of the species in the UVR (Neary and others 1996; Rinne and Stefferud 1996).

In the upper Gila River, five major, pre-designated study reaches were sampled between March and July of 1999, generally ranging in length from 150 to 300 m (492 to 984 ft) (Rinne and others 2005a, 2005b). Reaches were selected based on the primary objective of including a diversity of aquatic macrohabitats that have been demonstrated to be occupied by the native fish assemblage in the UVR (Rinne and Stefferud 1996; Sponholtz and Rinne 1997). These habitats were HGRs, LGRs, GRUNs, and POOLs of varying types. Initial physical descriptors of these habitat types are contained in Rinne and Stefferud (1997) and Sponholtz and Rinne (1997). Habitat types were initially designated qualitatively; however,

a direct readout digital flow meter, meter rule, and laser transit were used to define these units quantitatively after fish sampling. Only specific habitat data relative to spikedace and loach minnow abundance and distribution are reported herein. The other members of the native and nonnative fish assemblage and their habitats need additional analyses.

Fishes were collected in the upper Gila River by the same multiple sampling techniques that were used in the UVR. Direct-current, backpack electrofishing units were used to sample under debris, banks, and in HGR and LGR habitat. Normally, these two habitat types were sampled from up to downstream into a 5-m (16.4-ft), 3-mm (0.1-in) mesh bag seine. The GRUNs were normally sampled by seining from up- to down-stream with the same 5-m (16.4-ft) bag seine. All fishes collected in each unit were enumerated, measured, and returned alive to the same reach of stream. In the event of large numbers of individuals of a species, after 50 or more were measured, individual fish were enumerated only.

Depths were measured with a meter rule at 5-m (16.4-ft) interval transects in each macrohabitat. Velocities were estimated with a direct readout current meter at these same sites. Gradients were measured with a laser unit, and substrate was quantified by pebble count methodology (Bevenger and King 1995). Presence and absence data for the two species were compared and analyzed by nonparametric, Mann-Whitney tests to determine if a significant difference was present between these two data sets for both species.

Spikedace primarily (80%) occupy GRUN habitats in the UVR (Neary and others 1996; Rinne and Stefferud 1996) and secondarily in LGRs over gravel-pebble substrates. The species is often found on the UVR in greater abundance in “shear zones” where two riffle areas converge and form eddying currents (Rinne 1985b, 1991b, 1992, 1999a). Rinne and Deason (2000) reported the same habitat occupation by spikedace in the upper Gila River. The species was most common in average gradients of 54%. In 1994 in the UVR, GRUN habitats comprised about 40% of all habitats sampled. This percentage is near identical to that in the upper Gila in 1999 (Rinne and Deason 2000). Presently, only about 25% of habitats sampled in the UVR are GRUNs. Accordingly, there has been a about a 40% reduction in the well-documented optimal habitat for the species in the UVR over the past decade. Spikedace occupied waters that averaged 25 cm (9.8 in) deep with about 47 cm s⁻¹ (18.5 in s⁻¹) current velocity over gradients of approximately 1%. The gradients where spikedace were present and the habitat occupied provided the only significant ($P = 0.014$) physical habitat descriptor, as determined by a Mann-Whitney test. This rare species occupied substrates of sand-gravel composition in the upper Gila River (table 9.5). Availability of substrate components was nearly identical in the two rivers.

Spikedace appear to be broadly adaptable or perhaps broad in their habitat utilization. For example, mean water column velocity and depth where spikedace were present and absent were not statistically different ($P > 0.05$). The pelagic behavior of spikedace has previously suggested that mobility (Propst and others 1988) may be a rationale for a lack of habitat fidelity. Habitat use by this species does appear, however, to be delineated best by gradient or slope estimates. Most spikedace in the upper Gila were captured in GRUN and LGR habitat types over sand-gravel substrates. Rinne and Stefferud (1996) reported that 90% of all spikedace ($N = 904$) occupied GRUN habitats (mean velocities of 22 and 48 cm s⁻¹ [8.7 and 19.9 in s⁻¹], respectively) in the UVR. Similarly, Neary and others (1996) documented that 80% of spikedace that were captured in the UVR occurred in GRUN habitats, and the remaining 20% occupied LGR habitat containing 20 to 50% sand substrates.

Fish sampling in the upper Gila River in 1999 revealed gradients of respective aquatic macrohabitats that were very similar to those recorded for the UVR. Spikedace were most commonly collected in GRUNs (80%) and LGRs (20%, mean gradients = 0.54%; Rinne and Deason 2000). Spikedace were found in gradients ranging from 4.3 to 78 cm s⁻¹ (mean of 47 cm s⁻¹ or 18.5 in s⁻¹) and were most commonly found over sand-gravel substrates. There was a significant difference ($P = 0.0139$) in gradient between habitats where spikedace were present and absent, but this was not the case for depths and velocities.

Grazing Management Changes—UVR Corridor

Changes in grazing management on the river corridor confounded the question of which factor—flooding or nonnative fishes—was most influential in determining the status of all native fish, and specifically spikedace (Rinne 1985a, 1988). Since the spring of 1997, following continuous reductions in animal unit months over the previous decade, livestock grazing has been totally eliminated from the UVR corridor. As a result, riparian vegetation diversity and density have increased dramatically on the river corridor (Medina and others 2005). Neither native fishes nor the spikedace specifically have so far responded positively to this change in management although nonnative fishes have directly paralleled the increases in vegetation both on streambanks and within aquatic habitats. These correlations need additional study.

Conversely, there has been an inverse relationship between decreases in native fish populations, livestock numbers, and flooding and corresponding increases in nonnative fishes (Rinne 1999b). There are several factors interacting here (i.e., fishes, livestock, flooding, and vegetation). Lack of flooding has apparently been positive to vegetation distribution and abundance and to nonnative fishes, and it has been unfavorable to native fishes, including the spikedace. The dramatic increase in vegetation since 1997 due to the removal of grazing parallels an increase in nonnative fish abundance. The marked increase in two predatory nonnatives—smallmouth bass and green sunfish—may be a response, in part, to greater in stream cover associated with increased near stream and streambank vegetation density. These correlations and relationships could be coincidental; they have not yet been adequately studied or statistically tested. Rinne and Miller (2008) suggest that river and stream hydrographs characterized by periodic, variable instantaneous flow events are equally, and very likely more important, to native fish sustainability than is differing livestock grazing activity that results in variable changes in riparian/stream form and function. More research is needed to refine and delineate the relationship between native fish and grazing relationship (Rinne 1988, 1999b, 2000).

Interactive Factors and Management to Sustain Native Fishes

Notwithstanding known cycles of abundance (Minckley 1981; Propst and Bestgen 1986), the status of the spikedace and the entire fish assemblage in the UVR is perilous at best. Spikedace have declined in just a few years from 6 or 7% (1994 and 1996, respectively) of the native fish community to a level of non-detection. This decline coincided with an increase in base flow over the past two decades (Neary and Rinne 1998), and with a period of no flooding from

1996 to 2004. The combination of the predominance of nonnative fishes, the probability of entering a drought cycle, and the change in riverine habitat resulting from grazing management changes could lead to extirpation of one of only two large river populations of spinedace in the Southwest. Although statistically non-significant, the high probability that spinedace is extirpated from the UVR is cause for concern.

Of the three potentially interactive factors of flooding, nonnative fishes, and habitat change, only two can be addressed by management. Flood events are subject to the vagaries of the Southwest climate. Alternating cycles of drought and flood are the rule, not the exception. Data from the UVR (Stefferd and Rinne 1995; Rinne and Stefferud 1997; Brouder 2001) and the upper Gila River (Propst and Bestgen 1986, Propst and others 2008) and the conclusions of other documented effects of flooding on fishes (Minckley and Meffe 1987) show that native fish populations, including spinedace, appear to be positively correlated to flooding. In the UVR, spinedace have gone from common to very rare to absent from samples in just a few years during a period with no flood events and a drought or base flow condition (Rinne and Stefferud 1997; Rinne and others 1998).

Of equal importance in the UVR, as elsewhere in the West (Minckley and Deacon 1991) and Southwest (Rinne and Medina 1996; Rinne 1994, 1996), is that nonnative fishes consistently have resulted in a negative impact on native fishes. Whether by displacement or replacement (Douglas and others 1994), nonnatives apparently outcompete and prey upon native species (Minckley 1983; Meffe 1985; Rinne 1995b; Blinn and others 1993). Predation may indeed be the primary factor of replacement of native fish species by nonnatives (Clarkson and others 2005). At present, almost half of the individual fish inhabiting the UVR are nonnative. Combined with the lack of collection of spinedace for the past 12 years, this is cause for both concern and focused management by State and Federal agencies. In place, ongoing mechanical removal research should be continued. In addition, there should be consideration for experimental introduction of spinedace from the upper Gila within an adaptive management/research paradigm.

Finally, livestock grazing is a manageable land use activity. Anecdotally and coincidentally, livestock removal from the UVR in 1997 paralleled both the lack of flooding and the decline of spinedace in the river (fig. 9.15A). The indirect impact of livestock grazing on the watershed of the UVR presents a possible threat to spinedace sustainability. Although not discountable, the relative extrinsic impact of livestock grazing on the watershed, when compared to the predominance of an intrinsic, competitive, and predatory nonnative fish fauna in the UVR, becomes a moot point in the near term for management alternatives. Administrative/research collaboration should be attempted on selected reaches of river where both mechanical removal of nonnative fishes is ongoing and where it is absent.

Although complete removal of nonnative fishes using piscicides from the UVR is a highly improbable near-term management alternative, the objective of continually reducing and suppressing nonnative fishes should be the highest priority for management at present. Although immediate, complete removal of nonnative fishes is not administratively or logistically feasible, any efforts to suppress or reduce their abundance should be considered as a timely management alternative. Waiting for the next flood event to reduce nonnatives as a management alternative is not an appropriate strategy because of the time factor, the status of the spinedace, and the dominance of nonnative fishes in the UVR. However, continuing and synchronizing future nonnative fish suppression with

natural reduction in numbers by flooding (Rinne and Stefferud 1997) may be a viable management alternative.

Flooding is a natural disturbance event and, as previously discussed, may indeed be a prerequisite for native fish sustainability in the presence of nonnative fishes. Historically, floods and droughts have caused extreme fluctuations in the numbers and ranges of native fish, including the spokedace. After reduction of the entire fish community, re-colonization from refugia reaches has been possible historically in the absence of nonnative fishes and without many of the current mainstream dams and their alteration of flow regimes (Rinne 1994, 1995a, 1995b) and removal of fluvial connectivity.

The UVR has no significant mainstream impoundments or diversions below the start of perennial flow, only a nonnative fish assemblage. Further, with recent low drought flows and the removal of livestock, the riparian stream ecosystem of the UVR has become stabilized for the near term. Such stable aquatic conditions appear to be more favorable to nonnative fishes than natives in this river system. Even if nonnatives were absent in the UVR, the stability characterized by adequate and perhaps enhanced baseflow and the absence of floods (Neary and Rinne 1998) would promote increased instream and streambank vegetation and probably would be more favorable to native fishes. Nevertheless, until the next significant peak flow event occurs, nonnative fishes will continue to impact the spokedace—an impact that could result in extirpation of this threatened species from the Verde River system. If this happens, it would probably be favorable to attempt to experimentally restore spokedace and native fishes in general.

Even if a residual spokedace population exists and even if spawning does occur annually, the probability of survival of YOY spokedace is low given the excessive numbers of nonnative, predatory species present in the river (Stefferud and Rinne 1995; fig. 9.15A). An appropriate management alternative should be continued suppression of nonnative fish populations in the most upstream reaches of the UVR. Such action could help sustain native fishes and the spokedace locally (if present in future surveys) until the next flood event. Removal of grazing from the watershed or imposing limits on sport fishes are commendable extrinsic management strategies. However, both are cosmetic at best and pale in significance to intrinsic mechanical reduction in abundance of nonnative fishes, especially when coordinated with natural hydrologic reduction. Although mechanical, partial suppression of nonnative fish is not entirely effective, its continuation should be of high management priority in the total fish community (Rinne and Stefferud 1997).

Because of genetic considerations, the introduction of spokedace from other river systems has not been considered a viable management alternative (Tibbets and Dowling 1996). Isolation of the spokedace in the UVR from spokedace of the Gila River and Aravaipa Creek dates at least to completion of the Horseshoe and Bartlett dams on the Verde in the 1930s. Analyses of spokedace populations in the Gila River system (Tibbets and Dowling 1996) suggest that the population of spokedace in the UVR is genetically distinct. Accordingly, no translocation of spokedace should occur unless there is unequivocal evidence that the spokedace has been extirpated from the UVR drainage. At what point does one reach an unequivocal conclusion—extirpation of the species? Despite extensive sampling in Eagle Creek, Arizona, spokedace were not collected for more than two decades, but the species was recently confirmed with great effort at a disturbed road-crossing site in this stream. However, none of these situations justify adopting a non-action, non-progressive, or pro-active approach to management.

Three major additional management strategies should be strongly considered and instituted immediately. The first is intensification of survey and monitoring activity to search for spikewater. Secondly, locating and evaluating possible natural refugia streams or artificially establishing such in the Verde Basin should be considered. Lastly, culture techniques must be developed immediately for the species. These last two items should be in place when the next flood event occurs, which is presumably when spikewater will reappear in samples.

As discussed at the outset of this synthesis, RMRS in Flagstaff, Arizona, commenced study of fish populations in the UVR in 1994 following major flooding (figs. 9.11, and 9.13). Spikewater were most abundant in the most upstream reaches of the UVR during initial sampling in 1994 when 428 individuals were collected at the seven long term monitoring sites (Stefferd and Rinne 1995; Rinne 1998). Total catch in 1995 following less marked flooding dropped to 72 individuals, increased in 1996 to 140 individuals total captured. The species disappeared from samples at all seven sites in 1997 and has not been captured in the last 11 years of sampling. The species was common at four of the seven sites in 1994, rare by spring 1996 and absent in 1997 to present.

A dozen individuals were captured in the vicinity of the Black Bridge site in 1997. Reports of a single individual being captured by an Arizona Game and Fish Department Survey in the vicinity of the 638 Road site, however, no photographs or a specimen are available to document the record. Accordingly, the species has been absent from samples for over a decade (1996 to 2008) and has an ever-increasing probability of being extirpated from the river. The recent 2004/2005 floods greatly modified and improved habitat for spikewater (Rinne and Miller 2006, 2008) and reset many reaches of the river to that of 1993 flooding. However, three years of sampling at the seven sites, surveys by (60% of all spikewater captured in 1994) have failed to collect the species. Because of the high probability of the extreme reduction, and perhaps extirpation from the Verde River, and the aquatic ecosystem fragmentation of rivers and streams in the Southwest (Rinne and Calamusso 2006), experimental stocking of spikewater should be strongly considered in selected reaches of the UVR in the context of Sedell and others (1990).

Research and Management Implications

Long-term databases on temporal and spatial fish assemblage distribution and abundance in Southwest rivers and stream are wanting. Aravaipa Creek is likely the most lengthy (25 years) and robust in Arizona. In New Mexico, the upper Gila River has the most lengthy (10 to 20 years) database of this nature (Rocky Mountain Research Station database, Propst and others 2008). The database spanning 15 years and 60 km (37.5 mi) of the UVR is the most robust and sustained in time and space in Arizona. Almost two dozen publications have been produced since 1995 on not only fish distributions but also habitat use, interactions, flow regimes and fish assemblages, mechanical removal to sustain native species, and summary papers on fish assemblages (Rinne and others 2005a). Several cycles of flood and drought flows have been studied and their relative impacts on both aquatic and riparian corridor habitats and fish assemblages are documented. Continued reduction of native fishes and the disappearance of a threatened fish species, the spikewater, have been documented. Further, these physical and biological dynamics in the UVR have been contrasted and

compared to the upper Gila River with the same native fish fauna, but different flow regimes (Rinne and Miller 2006).

The data collected and publications printed over the 15-year time span suggest that it is important at this point to assess results and suggest some specific research and management activities that should be undertaken to build on with the objective to both sustain and enhance the native fish resource in the UVR. Some of the main themes of research/management that should be investigated:

- (1) The relative roles of nonnative fishes, flow regimes, and management activities such as fisheries and grazing management in the riparian corridor and extrinsic land use activities on the watershed on native fish sustainability.
- (2) The efficacy and effectiveness of mechanical removal techniques of nonnative, invasive fishes for sustainability of native fishes.
- (3) The relative roles of predation/competition by nonnative fishes and other invasive species such as crayfish, Asiatic clams, and bullfrogs on native fish sustainability.
- (4) The relative roles of primary and secondary production levels in native fish sustainability.
- (5) Experimental restoration of spokedace to the UVR through introduction of stock from the Gila River.

Long-Term Monitoring/Research of Fish Assemblages

The fish/habitat data base for the UVR is the largest of its kind in Arizona on a major river. Data have spanned three flood cycles, a substantial drought, base flow cycle, marked changes in fish species and assemblages, and stream channel habitats, disappearance of the threatened spokedace, and livestock grazing removal. A similar database of 10 years duration is extant for the upper Gila River. Continued monitoring and future research should be put in place to build on, compare, and further analyze these databases in light of the differential response of fish and species assemblages in these two river systems comprised of the same native and nonnative fish assemblages. A prerequisite to sound management is the best data and science on which to base it. Of critical importance is the restoration of the one threatened species—spokedace—to the UVR.

Mechanical Removal of Nonnative Fish Species

Mechanical removal on the UVR has been in place for almost a decade. In this time the factors responsible for success and or the lack of it in terms of response of native fishes have been well defined. Modified and intensified effort was instituted in 2006 and continues to present. The response of native fishes in spring 2008 at two of the long-term sites within removal treatments indicates a positive response to removal efforts. Therefore, it is highly recommended that these efforts be continued in out years. Further, land managers, RMRS scientists, and fish resource management agencies should strongly consider collaborative efforts to test the response of these efforts relative to the presence and or absence of management activities such as livestock grazing. Removal efforts should be considered as prerequisite to possible future chemical removal of nonnative fishes in the uppermost reaches of the Verde River.

Predation, Competition, and Primary/Secondary Production

Because of the ever-increasing literature on the impact of direct predation by nonnative fishes and other aquatic invasive species this impact should be examined in the UVR. Laboratory studies such as those of Carpenter and Mueller (2008) suggest that juveniles of even smaller-sized nonnative fishes can have a potentially negative impact on native fishes. Studies have been conducted by the University of Arizona (Schade and Bonar 2004, 2005) to lay groundwork for future studies. Primary and secondary production studies have not been a component of the past 15 years of study. Their role as related to flow regimes have not been studied intensively enough to determine their role in the sustainability of native fishes in the UVR.

Spikedace Monitoring and Restoration

The threatened spikedace disappeared from samples at the seven long-term monitoring sites within a few years after floods in 1992/1993 and 1995. The species has not been collected in the UVR since 1996 at the long term sites and 1997 below the Black Bridge long-term site. There should be a continued effort to monitor upper reaches of the river for the presence of this rare, threatened species. Large populations of this species yet occur in the upper Gila River in New Mexico. There should be an increased emphasis on consideration and implementation of restoring the species to the UVR, as is a component of the Recovery Plan (USDI Fish and Wildlife Service 1990). Restoration efforts must be coordinated with mechanical removal efforts and land management activities. Calculations suggest that the spikedace species has a high probability of being already extirpated from the UVR. The combined monitoring of RMRS, Arizona Game and Fish Department, and the U.S. Fish and Wildlife Service over the past decade and over the upper 60 km (38 mi) of river have failed to collect the species. Collaborative efforts should be initiated and pursued by these same agencies to experimentally introduce the species in the Burnt Ranch and Black Bridge long-term study site reaches.

Summary and Conclusions

The primary intent of this chapter is to summarize these works and then offer plausible management implications and research recommendations of this voluminous information into a state-of-the-art publication delineating management implications on fishes, fish assemblages, fish habitats and biology, and hydrology and geomorphology in the UVR based on monitoring and research from 1994 to 2008. In summary, the intent is to delineate for land managers: (1) what is known, (2) what is not known, and (3) what are the future research needs to facilitate management to sustain native fishes in the UVR.

Fish have been collected at seven long-term sites on the UVR since 1993. There has been a distinct decline in native fish species since the year after the 1993 flood and a parallel rise in nonnative fish species. Mechanical removal of nonnative fishes was not of sufficient intensity to have a significant effect with only an annual approach. Extensive streambank and in-stream vegetation reduced efficiency of removal, and limited treatment reaches (1 km or 0.6 mi) did not preclude emmigration of nonnatives into other sites.

The threatened spikedace disappeared from samples at the seven long-term monitoring sites within a few years after floods in 1992/1993 and 1995. The species has not been collected in the UVR since 1996 at the long-term sites, and since 1997 below the Black Bridge long-term site. Calculations suggest that the spikedace species has a high probability of being already extirpated from the Verde.