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Hydrology, Geomorphology and Management: Implications for Sustainability of Native Southwestern Fishes

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Native southwestern fishes have declined markedly in range and numbers. The factors responsible for their decline are many and varied. However, the primary stressors to native fish assemblages in southwestern rivers and streams are habitat alteration and introduction of non-native species. We present data that compare the fish assemblages in two desert rivers—the Gila and Verde (Arizona-New Mexico)—over a period of 7–12 years, respectively. We also present data on hydrographs, broadscale and local geomorphology, and past fisheries, water, and land management activities. Peak flow, mean volume of flow, variability of flow, canyon-bound and broad alluvial reaches, dams, and introduced fishes are all either directly or indirectly related to fish assemblages in southwestern rivers and streams. We suggest that three primary influencing factors—two natural and one human induced (hydrograph, geomorphology, management)—are critical features in delimiting native fish assemblages. Conserving and sustaining native fish assemblages in these and other southwestern rivers and streams will require land managers to address all aspects of these three major influencing factors with administrative and legal mandates.

Keywords hydrology, geomorphology, native fishes, Southwestern USA

Introduction

In the southwestern United States, the native fish fauna is low in diversity and is comprised primarily (95%) of cypriniform (minnow and sucker) species (Minckley, 1973; Rinne and Minckley, 1991). All native species have declined in range and numbers in the past 50 years (Miller, 1961; Rinne, 1994, 1996). As a result, most of the native fauna is either federally or state listed (Williams et al., 1989; Minckley and Deacon, 1991; Rinne and Minckley, 1991). Spikedace (*Meda fulgida*) and loach minnow (*Rhinichthys [Tiaroga] cobitis*) are two of the currently listed native southwestern fish fauna. These two federally threatened species are restricted to the Gila River basin—Arizona and New Mexico—and have declined dramatically in range and numbers (Minckley, 1973; U. S. Fish And Wildlife Service, 1990a, 1990b).

Largely because of regional hydrology and extensively modified river systems, research and management for native southwestern fishes has been approached on a species-by-species

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basis (Rinne and Stefferud, 1998). However, efforts must continually be made to study and manage native fishes at the assemblage level (Rinne et al., 1998; Rinne, 2003a, 2005). Information on factors limiting this disappearing resource (Rinne and Minckley, 1991) is needed by land managers to manage and sustain the native fish fauna in the Southwest.

Commencing in 1994, we initiated research and monitoring designed to determine factors that influence fish assemblage structure in the upper Verde River, Arizona (Stefferud and Rinne, 1995). Studies of fish populations and their habitats and possible abiotic and biotic factors influencing both have been conducted over the past 12 years in this reach of river, Arizona (Rinne and Stefferud, 1996, 1997; Rinne et al., 1998; Rinne, 1999a, 2005). In spring 1999, similar efforts to study fish assemblages were initiated in the upper Gila River, New Mexico, from its headwaters in the Gila Wilderness to the Arizona-New Mexico border (Rinne et al., 2005a). The primary objective of the effort on the upper Gila was to establish long-term monitoring sites for fish and their habitats. A second objective was to obtain temporal and spatial estimates of fish assemblages employing sampling methods similar to those used on the Verde. A major objective of the research and monitoring was to compare fish assemblages in the two river systems based on spatial and temporal changes in the native and non-native components and relative to factors possibly influencing respective assemblages. Because of critical threatened and endangered species issues and their legal ramifications, the distribution and abundance of two threatened species—spikedace and loach minnow—were of special interest.

Rinne (2002) introduced briefly the topics covered in this article. In this article, we examine in greater detail the primary factors that influence fish assemblages in the southwest. This article will: 1) describe fish assemblages in time and space in both rivers; 2) compare species trends in time and space in the two rivers; 3) describe trends in distribution and abundance of the two threatened species—spikedace and loach minnow; 4) outline factors that appear to be influencing or delimiting fish assemblage composition in the two southwestern desert rivers; and 5) relate these factors to management and conservation of the native fish resource in the arid American Southwest.

Study Areas and Methods

The primary study areas for the upper Verde and Gila Rivers are shown in Figure 1. Seven established monitoring sites have been sampled since 1994 in the Upper Verde River (Figure 2a) (Stefferud and Rinne, 1995). Additional major reaches (II–IV) from the headwaters to the mouth (Figure 2a) were also assessed with previously collected information provided by the Arizona Game and Fish Department.

The five major sampling reaches within the Upper Gila River are shown in Figure 2b. Sample sites in both rivers ranged in length from 150 to 300 m and were selected to include a diversity of aquatic macrohabitats that are occupied by all Gila River basin native species (Rinne and Stefferud, 1996; Sponholtz and Rinne, 1997). These same habitats were resampled annually to standardize catches among years. The specific habitat types are high-gradient riffles (HGR), low-gradient riffles (LGR), glide-runs (GRUN), and pools (POOL). Because physical descriptors of these habitat types are reported in Rinne and Stefferud (1996) and Sponholtz and Rinne (1997), specific habitat data relative to fish abundance and distribution will only be summarized here. Gradients of the different habitat types were estimated using laser technology. Velocities were measured with a direct readout current meter, and depths with a meter rule. Substrate composition was estimated using the methods of Bevenger and King (1995).

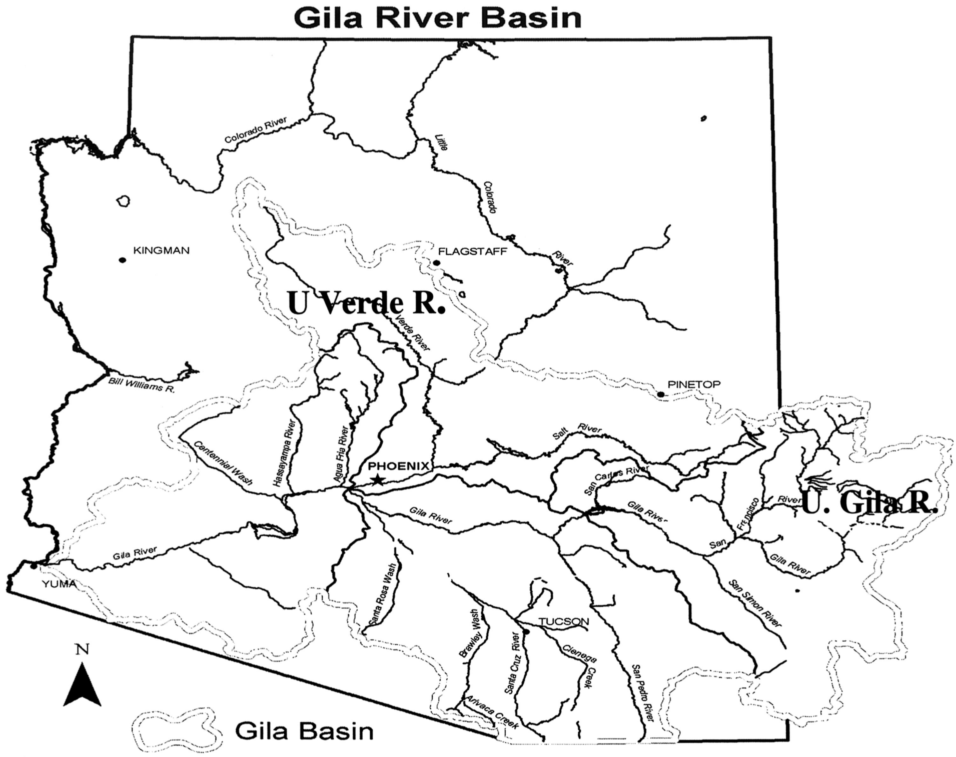


Figure 1. The Gila River Basin indicating the major study areas in the upper Verde River, Arizona and the upper Gila River, New Mexico-Arizona.

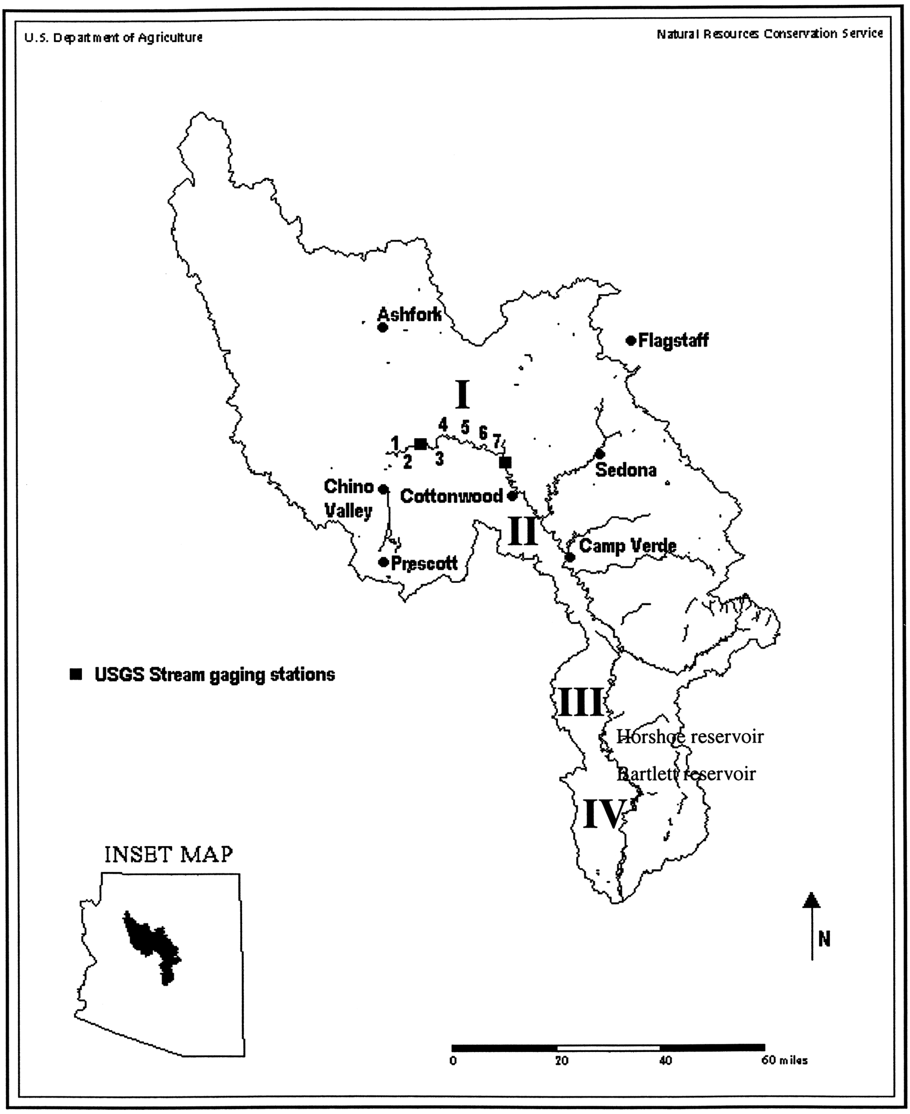
Fishes were collected by multiple sampling techniques depending on macro-habitat. Direct current, backpack electrofishing units were used to sample under debris, banks, and in riffles. In the case of HGRs and LGRs, shocking was conducted from upstream to downstream, and fish were collected into a 6-m, 3-mm mesh bag seine. Glide-runs were normally sampled by seining from up to downstream with the same bag seine. Deeper pools (>2 m) were trammel netted (30 m in length and meshes of 13-, 40-, and 80-mm mesh arrays) to sample for larger-sized (>30 mm) individuals. All fishes collected in each unit were counted, measured, and returned alive to the same reach of stream. Once 50 individuals of a species at a site were measured, all other individuals in a respective species were only counted. Hydrograph data are provided from the USGS web site www://water.usgs.gov/index.html.

Results and Discussion

Verde River

Fish. Total abundance of fish captured in Reach I in spring (April) from 1994 to 2005 has declined dramatically (Figure 3). The fish assemblage in this reach of river has changed from being predominantly (>80%) native from 1994 to 1996 to being dominated (>70%) by non-native fishes since 1997 (Figure 4). Similarly, downstream of Reach I to the mouth of the Verde River, non-native species increased and native species decreased (Figure 5).

Paralleling the overall decrease in native fishes, all six native species have declined markedly in abundance since initial sampling in 1994 (Table 1). Longfin dace (*Agosia chrysogaster*) numbered 1300 individuals in 1994, dropped to only a dozen individuals in 1995 (Table 1) and then increased to almost 300 individuals in 1996 before declining again to only 21 individuals among the seven sites in 1997 and a dozen in 1998. Only five individuals have been collected in the past 5 years of sampling at the seven monitoring sites.



(a)

Figure 2. (a) The seven established sites sampled in Reach I since 1994, indicating the four major reaches from the headwaters to the mouth of the Verde River. Horseshoe (H) and Bartlett (B) reservoirs are indicated, and b) map of the upper Gila River showing the five major reaches where sampling was conducted March–July 1999 through June 2005. Map modified from Natural Resources Conservation Service watershed map. *(Continued)*

(b)

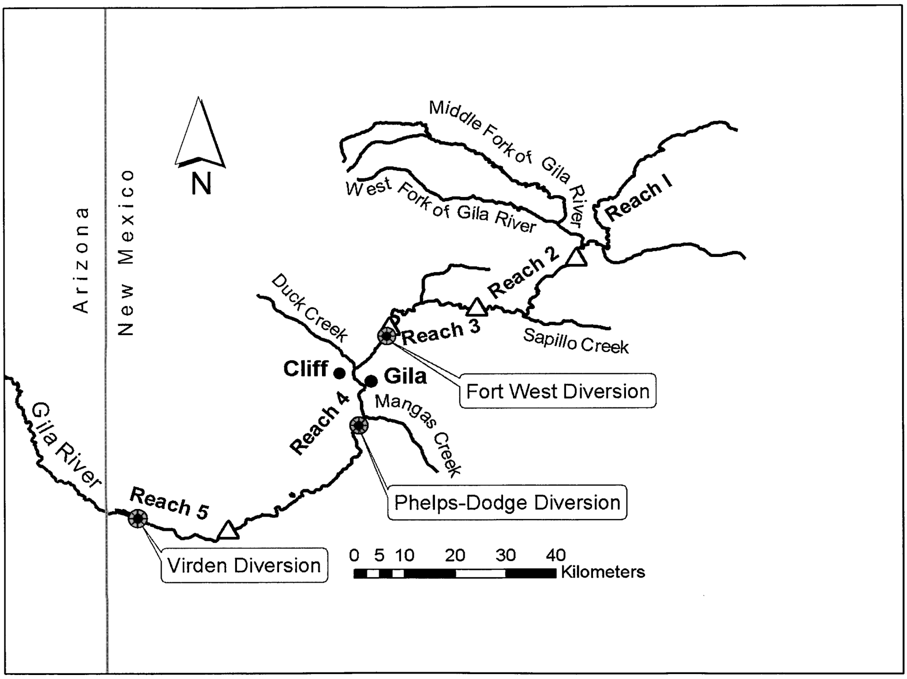


Figure 2. (Continued)

Similar to longfin dace, speckled dace (*Rhinichthys osculus*), another small-sized (<75 mm as adults) cyprinid, was most abundant in 1994 (171 individuals) before dropping 85% in 1995, more than doubling in 1996, and dropping to a single individual collected in 1997. A dozen speckled dace were collected in 1998, and only nine total have been collected between 1999–2005. None have been collected at the seven sites from spring 2001 to spring 2005.

As with longfin and speckled dace, abundances of the threatened spikedace, the last small-sized native species, were highest in 1994, dropped dramatically in 1995, increased slightly in 1996, and have dropped to zero at the seven established sites in all annual samples since 1997.

The three large-sized (>200 mm as adults) native species in the Upper Verde, desert sucker (*Catostomus clarki*), Sonora sucker (*Catostomus insignis*), and roundtail chub (*Gila robusta*), paralleled the smaller-sized species in temporal abundance (Table 1). Recruitment is poor in these three species and all have steadily declined in abundance since 1994 (Figures 6 a–c). Current (2005) numbers range from less than 1 to 3% of those recorded in 1994 following multiple, large flood events in winter 1992–1993 (see below).

By comparison, of the six non-native fish species, smallmouth bass (*Micropterus dolomieu*) and green sunfish (*Lepomis cyanellus*) have gradually increased in numbers between 1994 and 2003 before declining in 2004–2005 (Table 1). The other non-native species have fluctuated in abundances temporally. Mosquitofish (*Gambusia affinis*) increased markedly between 1997 and 2000, and except for 2004, has declined steadily in abundance since 2000 to the point of being absent in samples in spring 2005. Although numbers are still low, more (six individuals) young flathead catfish (*Pylodictus olivaris*) were collected in spring 2001 than in the previous 7 years of sampling; however, flathead and

Table 1
Fish assemblage structure estimated for the Upper Verde River, 1994–2005

Species	Year											
	1994	1995	1996	1997	1998	1999	2000	01	02	03	04	05
Native fishes												
Longfin dace	1319	12	282	21	12	2	1	2	1	1	0	1
Spikedace	428	72	149	0	0	0	0	0	0	0	0	0
Speckled dace	171	25	68	1	12	2	7	0	0	0	0	0
Desert sucker	2644	328	471	231	126	167	137	365	148	106	67	44
Sonora sucker	810	322	654	240	125	118	197	189	90	61	47	24
Roundtail chub	776	341	259	50	84	25	20	43	20	4	6	0
Nonnative fishes												
Smallmouth bass	14	10	32	35	66	104	48	170	211	150	57	13
Green sunfish	4	29	6	8	21	49	95	193	53	95	31	29
Yellow bullhead	31	29	9	40	33	15	22	36	19	21	16	2
Channel catfish	5	2	0	1	0	0	0	0	0	1	0	1
Flathead catfish	0	1	1	1	1	0	0	6	0	1	0	1
Common carp	23	6	13	19	9	4	15	15	4	3	4	10
Red shiner	1473	97	275	2238	1047	545	1594	1609	276	442	928	324
Mosquito fish	0	0	0	3	6	59	227	131	97	32	76	0
Percent native	82	86	85	19	22	29	15	19	28	17	15	16

channel catfish (*Ictalurus punctatus*) have been virtually absent in samples since 2003. Red shiner (*Cyprinella lutrensis*) has been the most abundant and cyclical non-native species in our decade of sampling on the Verde River. Samples in any year never contained more than 24 common carp (*Cyprinus carpio*).

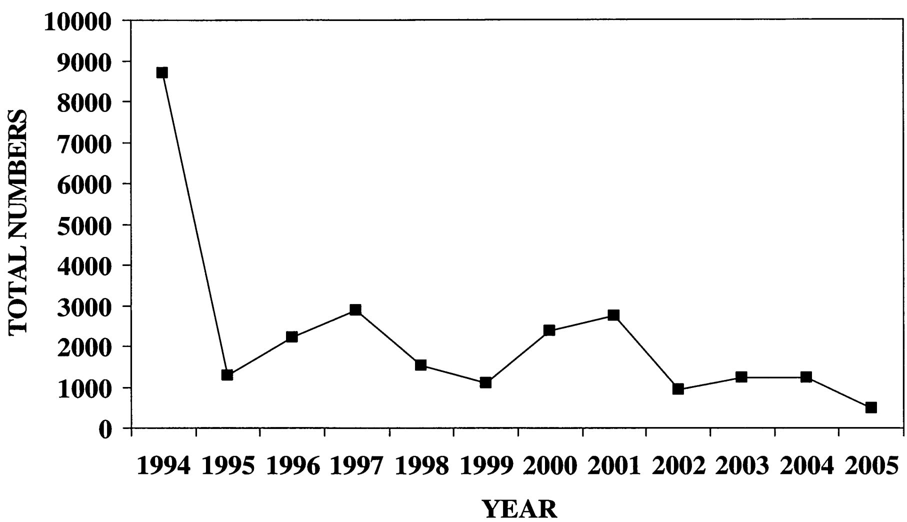


Figure 3. Total abundance of fish in Reach I, Upper Verde River, 1994–2005,

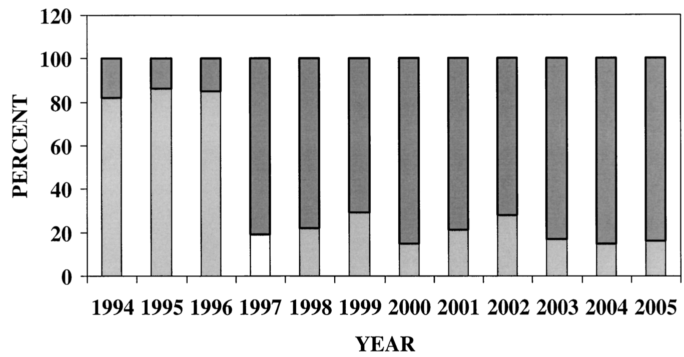


Figure 4. Relative components (%) of native (light bars) versus non-native (dark bars) species in the total fish assemblage in the Upper Verde, 1994–2005.

Habitat. Comparative flow statistics (Tables 4 and 5) and macrohabitat changes (Table 6) for the two rivers were calculated. The Gila River sustained much greater mean flow, flow variability, and peak or flood flows compared to the Verde River. Stream widths changed markedly in the upper Verde River between 1996 and 2000 resulting, in part, from livestock removal from the river and, in part, from a lack of flood events (Table 6). The channel became narrower, deeper, and streambank vegetation increased markedly (Rinne, 2006).

Gila River

Fish assemblages in the upper Gila River in the five major study reaches of river (Figure 2b) in 1999 are shown in Table 2. In Reach I, the Gila River headwaters in the Three Forks area, a single smallmouth bass was collected among the four sample sites. Similar to the Upper Verde River, desert and Sonora suckers comprised the major portion (60%) of the native fish assemblage. Speckled dace and roundtail chub were primarily (82%) collected at the West Fork of the Gila River site. Speckled dace were not collected in any of the four other major reaches in the mainstem Gila River. However, this species was abundant in Sapillo Creek at its confluence with the mainstem Gila River. All roundtail chub collected in the

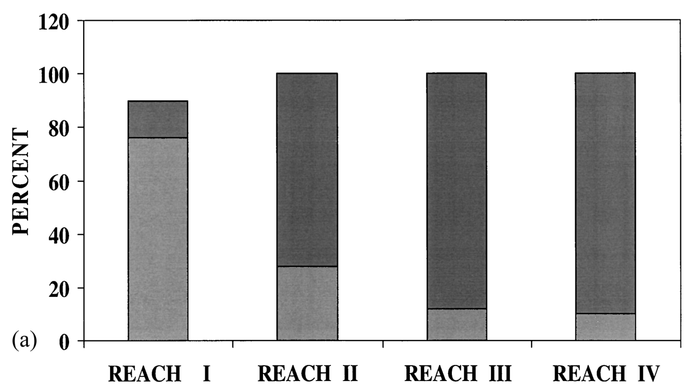


Figure 5. Relative abundance (% of total catch) of native (light bars) and non-native (dark bars) fish in the four major reaches of the Verde River Arizona: 1974–1997 (Arizona: Game and Fish records).

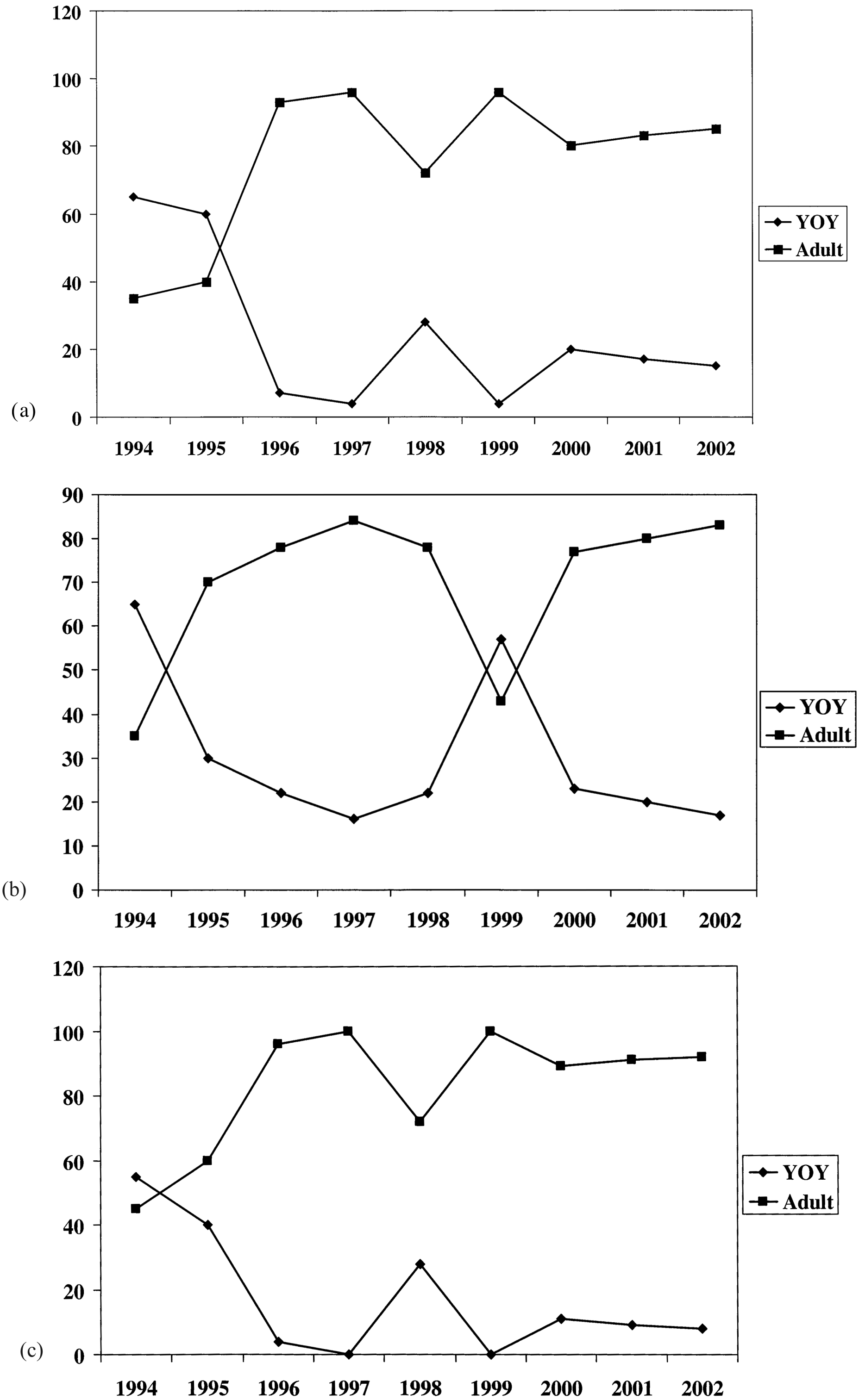


Figure 6. Relative proportions (%) of young-of-year (diamonds) and adults (squares) in the upper Verde River in autumn 1994–2002: a) Sonora sucker, b) desert sucker, and c) roundtail chub.

Table 2
Fish assemblage structure estimated at 17 sites in five major reaches (see Figure 2b) of the Upper Gila River, southwestern New Mexico, 1999

Site	PC	CI	AC	MF	TC	RO	GR	SMB	CAT	Other	Total
						Reach I					
W. Fk. Gila	3	42	2	10	0	67	53	0	0	0	167
M. Fk. Gila	32	63	5	0	37	3	23	0	0	0	163
E. Fk. Gila	165	43	6	69	11	0	0	1	0	0	295
Gila R.	95	24	0	0	27	7	0	0	0	0	153
Native/non-native ratio = 100/0									Total		778
						Reach II					
Smith Corral	60	3	21	0	0	0	0	16	3	2	105
Sapillo confl.	79	35	15	0	0	0	1	8	4	6	148
Sapillo Cr.	4	29	57	0	0	21	0	25	3	0	139
Seep Springs	4	0	3	0	0	0	0	4	0	11	22
Native/non-native ratio = 75/25									Total		414
						Reach III					
Brock Canyon	0	2	1	0	0	0	0	16	4	1	24
Watson Past.	32	15	17	0	0	0	2	12	1	10	89
Native/non-native ratio = 59/41						Total		113			
						Reach IV					
Riverside	9	0	3	22	0	0	0	0	1	0	35
Canyon Dam	39	5	5	189	13	0	0	1	1	0	253
Mangus Creek	354	0	86	280	20	0	0	0	0	0	740
Bird Area	32	25	8	50	1	0	0	0	0	0	116
Native/non-native ratio = 100/0									Total		1144
						Reach V					
Redrock	34	16	13	58	19	0	0	0	10	3	153
Nichols	6	1	0	14	0	0	0	0	6	0	27
Virden Diver.	72	24	243	12	1	0	0	1	5	331[1]	690
Native/non-native ratio = 59/41									Total		870
Totals	1010	327	485	694	129	98	79	84	38	365	3319

[1]All red shiners.

Species designations are PC, desert sucker; CI, Sonora sucker; AC, longfin dace; MF, spikedace; TC, loach minnow; RO, speckled dace; GR, roundtail chub; SMB, smallmouth bass, CAT, channel and flathead catfish; Other, all other non-native species such as sunfish, and bait species primarily comprised of red shiner (see footnote for Virden [Sunset] Diversion).

West Fork of the Gila River were taken in a single, large pool containing extensive woody organic debris.

The two threatened species, spikedace and loach minnow, were present in Reach I; however, spikedace were collected only at the West Fork and East Fork Gila River sites. No loach minnows were collected at the West Fork Gila River site; however, both loach minnows and spikedace were taken about 1.5 km downstream from the West Fork Gila River sample site.

In Reach II, native fishes still predominated (75%) at three of the four sites sampled in this canyon-bound Gila Wilderness reach of the Gila River near the mouth of Sapillo Creek. Desert and Sonora suckers again comprised the major component (64%) of the native fish assemblage; however, longfin dace was the second most common species to desert sucker and comprised 31% of the native fish assemblage. In contrast to Reach I, non-native species increased and comprised 20% of the total fish assemblage in Reach II. Further, spikedace and loach minnow were absent at all sample sites within this reach. Smallmouth bass (65%) was the dominant non-native species. Spikedace and loach minnow were absent at all sample sites in Reach II.

In Reach III, non-native species comprised almost 41 of the total fish assemblage. Similar to Reach II, no spikedace or loach minnows were collected at this outlet reach of the wilderness canyon before the reach transitions into the alluvial Gila River Valley near Cliff, New Mexico (Reach IV). Again, desert and Sonora sucker made up the largest component (73%) of the native fish community.

Overall, fish abundance in Reach IV increased markedly from the upstream two reaches (II and III). Desert sucker and Sonora sucker again made up the major portion (43%) of the total fish assemblage; however, spikedace and loach minnow combined comprised 49% of the native fish assemblage. Longfin dace (9%) was the only other native species collected. Non-native species were virtually absent in samples in this reach: only a single smallmouth bass and two yellow bullheads (*Ameirus natalis*) were collected.

Total fish abundance decreased slightly from Reach IV to Reach V, and the two native suckers comprised 43% of the native fish assemblage. Native fishes made up only 42% of the total fish assemblage largely because of the abundance of red shiner at the Virden diversion site. Spikedace abundance decreased dramatically (84%) and loach minnow decreased 40% from its abundance in Reach IV. To summarize 1999 samples, spikedace and loach minnow were present in Reach I, absent in Reaches II and III, most abundant in Reach IV, and declined markedly in numbers again in Reach V.

Although temporal distribution and abundance data at specific sites are not as extensive in the Upper Gila River, 7 years of data at five U.S. Bureau of Land Management and private land sites are currently available (Table 3). Overall fish abundance was variable at the five sites. Between 1999 and 2005, total numbers of each species increased and decreased variably. Of all five sites sampled over the 7 years, the non-native component of the fish assemblage comprised greater than 10% of the total fish assemblage on only seven occasions. Spikedace and loach minnow were only present or most abundant in the initial year of sampling at Bennett Place, were most consistently abundant at Fred's Place and Redrock, and became very low in numbers or absent (2003–2005) in samples collected at Nichols Canyon and Virden Diversion. Loach minnow did reappear in samples at Nichols in 2005.

In summary, in the Upper Verde River, both total fish numbers and numbers of native fishes decreased over the 12 years of sampling. Native species decreased steadily in numbers and the native component of the fish assemblage decreased below 20% from 1997 to 2005. Spikedace became absent in samples at the seven sites in 1997, and longfin and speckled dace were rare-to-absent at the same time. Conversely, in 1997 the non-native component surpassed the native component and has maintained itself at 80% or greater. The native component of the fish assemblage also decreased downstream in the four major reaches of the Verde River. In the upper Gila River, total fish and numbers of natives were most

Table 3

Changes in fish assemblages at five U.S. Bureau of Land Management and private land long-term monitoring sites sampled from 1999 to 2005 in the Upper Gila River, NM

Loc.	Year	PC	CI	AC	MF	TC	R0	CAT	Other	Total
					(Reach III)					
Bennett Place	1999	109	2	46	8	30	1	0	0	196
	2000	20	1	0	0	8	0	0	0	29
	2001	5	92	14	0	1	0	0	0	112
	2002	0	0	0	0	0	0	0	0	0
	2003	0	33	0	0	0	0	2	90	125
	2004	3	802	0	0	0	0	2	29	836
	2005	3	120	96	0	2	0	1	6	221
Fred's Place	1999	9	1	22	41	14	0	0	0	87
					(Reach IV)					
	2000	33	121	63	5	48	0	1	0	271
	2001	12	215	5	11	5	0	0	5	253
	2002	41	1070	131	19	40	0	0	69	1307
	2003	0	1923	114	4	5	0	0	1	2047
	2004	84	220	41	50	51	0	0	4	450
	2005	444	99	1274	113	76	0	1	0	2007
Redrock	1999	34	16	13	58	19	0	10	3	153
					(Reach IV)					
	2000	9	287	504	9	10	0	15	0	879
	2001	45	44	35	1	11	0	2	5	143
	2002	100	60	641	42	8	0	34	19	967
	2003	62	8	1	0	1	0	7	87	166
	2004	5	0	8	0	0	0	5	59	81
	2005	41	19	127	10	4	0	7	41	251
Nichol's Canyon	1999	6	1	0	14	0	0	6	0	27
					(Reach V)					
	2000	3	481	262	5	0	0	1	41	793
	2001	19	275	79	9	1	0	1	25	409
	2002	75	83	194	5	0	1	127	26	510
	2003	128	19	7	0	0	0	2	33	189
	2004	2	0	4	0	0	0	6	74	86
	2005	4	0	91	0	3	0	43	10	153
					(Reach V)					
Virden (Sunset) Diversion	1999	72	24	243	12	1	0	1	331	684
	2000	1	13	29	1	0	0	49	11	104
	2001	19	33	41	17	0	0	0	12	122
	2002	39	43	34	5	0	0	6	7	134
	2003	25	4	3	0	0	0	2	9	43
	2004	0	0	2	0	0	0	2	8	12
	2005	206	2	92	0	0	0	15	0	315

Species designations are the same as give in Table 2.
CAT is for all catfishes and OTHER includes all other non-native fishes as defined in Table 2.

abundant in the uppermost reach in 1999, declined in numbers through the Gila Wilderness canyon (Reaches II and III) before increasing in abundance in Reach IV (the Gila/Cliff Valley) (Figure 2b). Native fish abundance declined in both abundance and percentage of the total fish assemblage in Reach V. The threatened spikedace and loach minnow were present in Reach I, disappeared in the Reaches II and III in the wilderness canyon, reappeared and become very abundant in Reach IV before declining again in the lowermost Reach V.

Practical Applications for Resource Managers. At the broadest scale, two major categories of factors affecting native fish and their habitats must be considered: 1) natural and 2) anthropogenic or human-induced influences. Because both types of factors interact and have cumulative effects, interpreting their relationships and relative effects on fish, their habitats, and their sustainability is difficult at best. However, managers must understand and manage native fishes not only from an administrative and legal perspective, but equally important, within a context of natural processes and functioning of southwestern river systems (Rinne, 2002, 2003a; Rinne et al., 2004; Medina et al., 2005). Further, they must consider human land and riparian management activities and their subsequent influences relative to these natural factors. By doing so, the likelihood that this valuable natural resource will be sustained and enhanced increases. We suggest there are several guiding principles or generalizations that land managers should understand and consider in efforts to conserve and sustain the native fish assemblages in southwestern desert rivers.

1. Hydrographs of southwestern desert rivers are fundamental to delimiting fish assemblage structure. Based on USGS data from the Paulden gage on the Verde River and the Gila gage on the Gila River, hydrographs are very different between the two rivers (Tables 4 and 5). First, mean annual streamflow in the Gila/Cliff Valley reach is almost four times that of the Upper Verde River. Second, the range of mean monthly discharge varied only 0.57 m^3/sec in the Verde River compared to 19 m^3/sec in the Gila River, or 20 times

Table 4
Flow statistics (hydrographs; m^3/sec) for the Verde and Gila Rivers at USGS Paulden and Gila gages between 1993 and 2005 comparing variability and peak flows between the two rivers

Parameter	Verde	Gila	Comparative Factor: Gila × Verde
Mean annual discharge	2	9	4×
Monthly discharge			
Range	.57–1.14	84–20	1–20×
Mean			
Winter	2.4	7.0	3×
Spring	0.8	4.0	5×
Summer	.94	3.4	4×
Autumn	.94	3.7	4×
Instantaneous peak discharge			
>143 (5000 cfs)	11	23	2×
>285 (10,000 cfs)	4	12	3×

Table 5
Annual maximum instantaneous peak flow (m^3/sec) comparisons in the Upper Verde and Gila Rivers, 1993–2005. Data are from the U.S. Geological Survey's Paulden and Gila gages

Year	Verde Rive	Gila Rive
1993	630	405
1994	5	12
1995	113	476
1996	30	72
1997	6	519
1998	17	60
1999	51	79
2000	43	86
2001	17	37
2002	43	38
2003	25	6
2004	329	21
2005	334	369

greater in the Gila than in the Verde River (Table 4). Third, mean monthly stream flows for the four seasons averaged three to five times greater in the Gila River than the Verde River. Fourth, instantaneous peak discharges in the Gila River, greater than 143 m^3/sec (5,000 cfs) and 285 m^3/sec (10,000 cfs) between 1993 and 2005, were twice to three times those in the Verde River. Finally, between 1993 and 2005, only in 5 of the 13 years did the Gila River have a maximum peakflow of less than 57 m^3/sec (1,200 cfs) (Table 5). By comparison, the Upper Verde River was less than the 57 m^3/sec peak flow level in 9 of those 13 years. Furthermore, most (8 of 9) of these low (<1,200 cfs) flows in the Verde River occurred between 1994 and 2003 compared to 4 of 5 in the Gila occurring between 2001 and 2004.

Further comparison of instantaneous peak flows (an indicator of level of flooding) in the two rivers since 1993 is instructive (USGS records) (Table 5). In 1993, peak flow at the Paulden gage (Figure 2a) was 630 m^3/sec. In 1995, maximum instantaneous peak flow was almost 114 m^3/sec at this gage. Peak flows in the Verde River in the decade between 1994 and 2003 have exceeded 75 m^3/sec only once since 1995. By comparison, peak flows in the Gila River exceeded 75 m^3/sec four times in this same decade and exceeded 400 m^3/sec in both 1993 and 1995. In contrast to the Verde River, peak flows from storms generated by Hurricane Linda in September 1997 exceeded 513 m^3/sec, which was the 4th highest peak flow ever recorded at the Gila gage since records began in 1928.

We suggest that instantaneous peak flows or the flood event component of the hydrograph partly accounts for the differences in fish assemblage structure in the two rivers. Stefferud and Rinne (1995) and Rinne and Stefferud (1997) partially substantiated this relationship for the Verde River and Minckley and Meffe (1987) did the same for other streams in the southwest. Both rivers sustained substantial floods in the mid 1990s; however, none have occurred in the Verde River between March 1995 and September 2004. The Gila River has a more variable and greater output of stream flow (volume) than the Verde River (Table 4). We suggest the two hydrological variables—variability and volume—are equally or more important than instantaneous peak flows in influencing fish assemblages in desert rivers. Combined, all three factors (i.e., peak flow, variability of flow, and volume of flow) very likely explain the lack of non-native species in three of the five reaches in 1999 in the upper Gila River (Table 2) and the sustainability of this fish assemblage component between 1999 and 2005 (Table 3).

In summary, based on hydrologic data from the two rivers, peak or flood flows appear to have a pronounced, positive effect on most of the native fishes. However, the variability and differing flow volumes (Table 4) between the two rivers appear to influence microhabitats and fish assemblages (see below). That is, more variable hydrographs and greater flow volume sustain native fishes over non-natives between periodic flood events (Rinne, 2004). It is notable that since 1993, large (>400 m^3/sec) floods have occurred every other year up to 1997 in the Upper Gila River. Between 1998 and 2004, only lower peak flows (<86 m^3/sec-3000 cfs) have occurred and yet non-native fishes have generally increased at three of the five long-term sites (Table 3).

Similarly, by 1997, non-native fishes became the dominant component of the total fish assemblage in the Upper Verde River (Rinne et al., 1998; Rinne, 1999a; Rinne, 2006). The last flood event greater than 86 m^3/sec was in 1995. This desert river has been in drought and low peak or lack of flood flows since that time. At the time of this writing, no threshold of discharge that might stimulate reproduction and native fish increases could be offered (Rinne and Stefferud, 1997; Rinne, 2003a). The relative role of the hydrograph in structuring southwestern fish assemblages can only be better understood by continuing to monitor fish assemblages and hydrographs in the Verde River (and Gila River) until the next significant flood event. Defining a significant flow requires observations of fish assemblage response relative to the size of the event.

2. Geomorphology on two different scales is basic to sustaining southwestern native fishes. Broadscale geomorphology. Platts (1979) suggested geomorphology was an important determinant of fish community structure. On a localized, reach scale, specific habitat of fishes has frequently been reported (Armantrout, 1981). Temporal-spatial variations in distribution and abundance of spikedace and loach minnow in the upper Gila River are evident (Tables 2 and 3). Neither species was collected in the lowermost extent (Reaches II and III) of the canyon-bound reaches of the Gila Wilderness portion of the upper river, yet comprised significant proportions of the native fish assemblage in Reaches I (20 %) and IV (52%). No obvious differences in habitat availability for these two species were evident among these reaches (Rinne et al., 2005a).

Map estimation of gradient of the two rivers along their entire course sampled appears identical (0.5%). However, in Reach III of the canyon-bound segment of the Gila River, mean gradient was calculated at 0.8%. By comparison, the broader alluvial reaches (IV and V) were calculated to be 0.4% and 0.3% in mean gradient, respectively. Because of very specific habitat preferences of the native fishes (Rinne and Stefferud, 1996; Sponholtz and Rinne, 1997; Rinne, 2003a), smaller scale, localized geomorphic/fluvial, macro-habitat influences in these rivers are very basic to fish abundance and distribution. That is, aquatic macrohabitats (e.g., HGR, LGR, GRUN, and pools) are very directly linked with dispersion and abundance of the native fishes. Reduction of gradient by 50% or more in Reaches IV and V compared to Reach III results in the probability of more LGRs and GRUNs and may be significant in determining fish assemblages. Rinne and Deason (2000) documented these two habitat types as optimum for spikedace. Calamusso and Rinne (2002) noted distributional changes in one native sucker in New Mexico relative to slight changes in stream gradient.

Notable are both the relative abundance of non-native species in general and the presence of larger (>300 mm), predatory catfishes in deeper (>2 m) pool habitats in the Gila Wilderness reaches (Reaches II and III) and at sites in Reach V, a canyon-bound reach below the lower Gila Box. The presence and piscivorous habits of the non-native species must certainly affect both the presence and abundance of native species such as the roundtail chub and Sonora sucker. Only a single, small (66 mm, TL) chub was collected in Reach II and two were collected in Reach III (Table 2). Both the overall geomorphology and that reflected in local aquatic microhabitats were probably partly responsible for the low numbers of native fishes. This is consistent with native fish distribution and abundance relative to specific habitat featuress (e.g., velocity, substrate, gradient) (Rinne and Stefferud, 1996; Rinne and Deason, 2000).

The influence of pools on fish assemblages is best illustrated by data from the Upper Gila River (Rinne et al., 2005a). For example, based on habitat data in the canyon-bound middle reaches (II and III), the relative number of pools is greater than in the alluvial valley reaches. Further, removing pools from the analysis of fish assemblage structure dramatically and positively alters native/non-native fish ratios to the benefit of natives (Rinne et al., 2005a). In 3 of the 5 years of sampling pool habitats at the Redrock site (Reach V), a large number of catfish including large channel (*Ictalurus punctatus*) and flathead catfishes were captured. An attendant reduction of native fishes in pools containing these large predators plus and increase in smaller predators (sunfish and smallmouth bass) during successive years of sampling strongly suggests their negative impact on native fishes.

Finally, narrowing and deepening of the instream channel in the Upper Verde River (Table 6) effectively creates or mimics "pool type" or deeper water habitats. Channel confinement by vegetation has resulted from removal of livestock grazing in 1997 and a lack of significant flooding since 1995 (Rinne, 2006). Narrower channels have produced habitats better suited for the larger, non-native predatory species such as smallmouth bass. Narrowing and deepening of instream aquatic habitat has been documented to be beneficial to salmonid species (Platts, 1991). However, despite two of the larger native species (roundtail chub and Sonora sucker) being pool inhabitants, the other four species are more shallow water riffle and glide-run inhabitants (Rinne and Stefferud, 1996). These two habitat types (LGR and glide-run) are rare in the Upper Verde River. By contrast, they are ubiquitous in Reach IV or the alluvial Gila-Cliff Valley.

In summary, canyon bound reaches have a higher probability of the occurrence and greater depth of pools, which are more optimal habitat for large, non-native predators such as catfish and smallmouth bass. In contrast, broad alluvial valleys sustain fewer and shallower (<2 m) pools due to the dynamics of hydrology and bedload movement and sorting that tend to aggrade rather than degrade stream channels—conditions more favorable to some native fish species. Rinne and Deason (2000) documented strong selection of subtrate types in the Upper Verde Rvier by spikedace and loach minnow (Rinne and Stefferud, 1997)

Table 6

Comparison of physical habitat change (width and depth in meters) between the Burnt Ranch and Perkinsville sites in 1994, 2000, and 2005

	1994		2000		2005	
	Burnt Ranch	Perkinsville	Burnt Ranch	Perkinsville	Burnt Ranch	Perkinsville
Width	6.3	6.0	3.6	2.9	10.0	12.0
Depth	.26	.19	.35	.38	.24	.20

Specific aquatic macrohabitats. Aquatic macrohabitat types for the two rivers in 1999 were described by Rinne et al. (2005a) and Rinne and Deason (2000). Several differences were notable. First, calculations revealed there was an almost complete lack of HGR habitats (90 cm/sec or greater mean velocity) in the Upper Verde River compared to the Gila River, where HGRs comprised a little less than a third of all the habitats sampled. The lack of this habitat type and the fact that HGRs are optimal for loach minnow may be responsible, in part, for the absence of loach minnow in the Verde River. Second, during random sampling of study reaches, there was about half as many pools sampled in the Gila River compared to the Verde. Low-gradient riffles and GRUNs were similarly represented between the two rivers. Finally, in Reaches III and IV of the Gila, HGRs comprised a lower percentage (<25%) of the habitats sampled. Pools were evenly distributed throughout all sample reaches on the Gila River; however, deeper pools (>2 m) were rare in Reaches IV and V. Low-gradient riffles and GRUNs, habitats in which spikedace are normally captured (Rinne and Deason, 2000), comprised almost half of habitats sampled in Reaches I and II and in a majority of all habitats in Reaches IV and V (60% and 67%, respectively). The lowest percentage (37%) of these combined habitat types was in Reach III.

Not only is habitat type important, but also habitat diversity and physical location in a reach of river affect fish assemblages. Rosgen D-type channels (Rosgen, 1994, Rinne, 2003b), characterized by stream braiding, are currently viewed as an indication of "instability" and "increased sediment loading in stream channels." Nevertheless, these channel types appear more favorable to native fishes in general, and especially to the two threatened species—spikedace and loach minnow. However, more complete analyses of the relationship of D channels and native fishes are needed.

In summary, a mosaic of interdispersed HGRs, LGRs, and GRUNs, accompanied by a lack of pools (especially deeper, >2 m, pools), appears optimum for the native component of the fish assemblage (Rinne, 2003b). To recap, deep (>2 m) pools provide more optimum habitat for non-native predatory species such as smallmouth bass and catfishes. By contrast, a lack of such habitats reduces the abundance of these large-sized, piscine predators.

3. Management activities affect fish assemblage structure in southwestern rivers. Grazing Management. Coinciding with the current dominance of non-natives in Reach I in the Upper Verde River has been the removal of livestock grazing in 1997 (Rinne, 2006). Since that time, riparian and instream vegetation have increased dramatically (Rinne, 1999a; Medina and Rinne, 1999; Medina et al., 2005; Rinne, 2003b). We suggest that the resulting marked increase in instream and stream bank vegetation and narrowing and deepening of the channel mentioned above provide better habitat for cover-seeking species such as smallmouth bass and green sunfish (Pflieger, 1975). How these changes in grazing practices affect native versus non-native cypriniform fish and their habitats is not fully understood (Rinne, 1999a, 2000). These relationships need to be better defined with more specific, comparative studies of fish habitat relative to grazing on the Verde, Gila, and other rivers in the southwest. Only a preliminary study has been completed on the Verde River (Rinne and Neary, 1997) and none has been conducted on the Upper Gila River. Further studies are needed to determine if a connection exists between grazing, specific fish habitat, and fish presence and abundances (Rinne, 1999b). For example, controlled experiments could be conducted where 1–2 km reaches of the Upper Verde could be selectively grazed, and the fish communities of grazed and nongrazed reaches could then be compared.

Fisheries Management. Over the past century, fisheries management in southwestern rivers has introduced many non-native sport species (Rinne, 1996; Rinne et al., 2004; Cowley, this volume). For example, about 100 species of non-native fish have been introduced into the waters of Arizona since the late 1800s and half of these species have become established (Rinne, 1994). Hundreds of stocking events involving millions of individual fishes have occurred on the Verde River (Rinne et al., 1998). Except for seasonal stocking of trout in the reach of river near Cottonwood, Arizona, most stocking in the river proper has ceased and occurs in reservoir environments for sport fishing enhancement.

Since 1994, smallmouth bass has increased steadily in samples in the Upper Verde River (Rinne, 2001) (Table 1). The presence of many (ca. 40%) age 1 smallmouth bass in the spring 1999 sample indicated favorable habitat and reproductive conditions for this piscivorous non-native species. Furthermore, non-native fish species have increased steadily in abundance in the Upper Verde River, in part, because of the extensive stocking events over the past 60 years (Rinne et al., 1998). The increased abundance of juvenile flathead catfish in Spring 2001 samples is cause for alarm in the Upper Verde River. Prior to 2001, only four individuals were collected (Table 1). By contrast, six young flatheads were collected in 2001 alone. This species has completely replaced native fishes in the Salt River (Kirk Young, Arizona Game and Fish Department, Phoenix, personal communication) above Roosevelt Lake.

By comparison, stocking events have been limited in the Gila River relative to the Verde River. Lack of sustained introductions in combination with the hydrology and geomorphology of the Gila River have precluded greater abundance of non-native, sport species in all reaches but those in the Gila River wilderness, canyon-bound reaches. We postulate that this increased abundance of large predatory fish in these reaches largely results from the presence of deeper (>2 m) pools formed through the interactions of flood flows and canyon walls that result in increased degradation in these reaches.

Hydrological management (dams and diversions). The U.S. Bureau of Reclamation dam building era commenced with Roosevelt Dam in 1911 on the Salt River (Rinne, 1975; Rinne, 2003b; Rinne et al., 2005b). Neither Reach I of the Upper Verde River nor the Upper Gila River has a major dam impounding significant volumes of water. The Upper Verde River (Reach 1) has only Sullivan Dam near Chino Valley that impounds no permanent pool and one minor water diversion at Perkinsville (Figure 2a). By comparison, the Upper Gila River sustains three large diversions, one each in Reaches III, IV, and V (Figure 2b). During the Spring 1999 sampling, flows were very low (<6 m^3/sec) but the Phelps Dodge Diversion in Reach IV) (Figure 2b) did not dry the river. However, the Sunset and Fort West Ditch diversions (Figure 2b) completely removed all flow from the river channel in summer 1999 and 2000.

Non-native fish distribution and abundance are affected directly by dams and diversions (Rinne, 1994, 1996; Rinne et al., 2005b). Mainstem dams are absent in upper reaches of both rivers and do not play a major role in delimiting fish assemblages in these uppermost reaches of the two rivers. However, the effects of mainstream dams downstream (Figure 2b) on native fishes in the Upper Verde have been documented (Rinne et al., 1998). Principally, the alteration of natural flow regimes from stochastic to regulated flows appears to be more beneficial to non-native fish. By comparison, in the Upper Gila, reduced in-stream flow or complete drying as was observed in spring 1999 below the Sunset (Virden) Diversion, obviously has a marked impact on the entire fish community.

Summary and Conclusions

The two rivers examined tell two different stories of southwestern desert river fish assemblages. We hypothesize that the interactions of hydrology and geomorphology in combination with human activities, especially past fisheries management practices, explain these differences. That is, lower, stable base flows during a time of drought (1996–2004) in the Upper Verde River have been favorable for non-native fishes (Rinne and Miller, 2006) (Table 5). In contrast, in the Upper Gila River, non-native fish, although present, have not increased in abundance because of flow regimes that result in a lack of aquatic vegetation, shallower waters and a general lack of pools (Rinne, 2006). Furthermore, monitoring of fish communities in the Verde and Gila Rivers and comparing these assemblages to corresponding hydrographs and human-induced changes in stream dynamics and composition, should continue and be expanded. Other rivers in the southwest should also be studied to test our hypotheses. We contend desert river systems are complex and very dynamic. Flow alteration and introduced fishes as major stressors to native fish assemblages in North America have been documented (Rinne et al., 2005b). Using simple linear, one-to-one relationships will not likely give land managers the answers needed to align management to sustain native fishes for perpetuity.

In a management context, the human-induced factors (e.g., fisheries management decisions, hydrologic modifications, grazing, and other landscape uses) can be addressed most directly relative to native fish sustainability. Geomorphic habitat at the reach scale can be affected by land management activities. In contrast, natural, broad-scale geologic features (i.e., narrow canyons versus broad alluvial valleys) cannot be feasibly altered through management. Hydrographs may be influenced by landscape and watershed uses. In summary, the interaction of natural factors and anthropogenic activities will continue to affect fish assemblages in aquatic habitats in the Southwest. Restricting future introductions of non-native fish in nearly pristine rivers and streams, restricting flow modification practices such as damming, diversions, or groundwater pumping, and ensuring that grazing practices are compatible with the goals of fisheries managers are the primary management strategies that will increase the probability that native fish assemblages will be sustained in southwestern rivers and streams.

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