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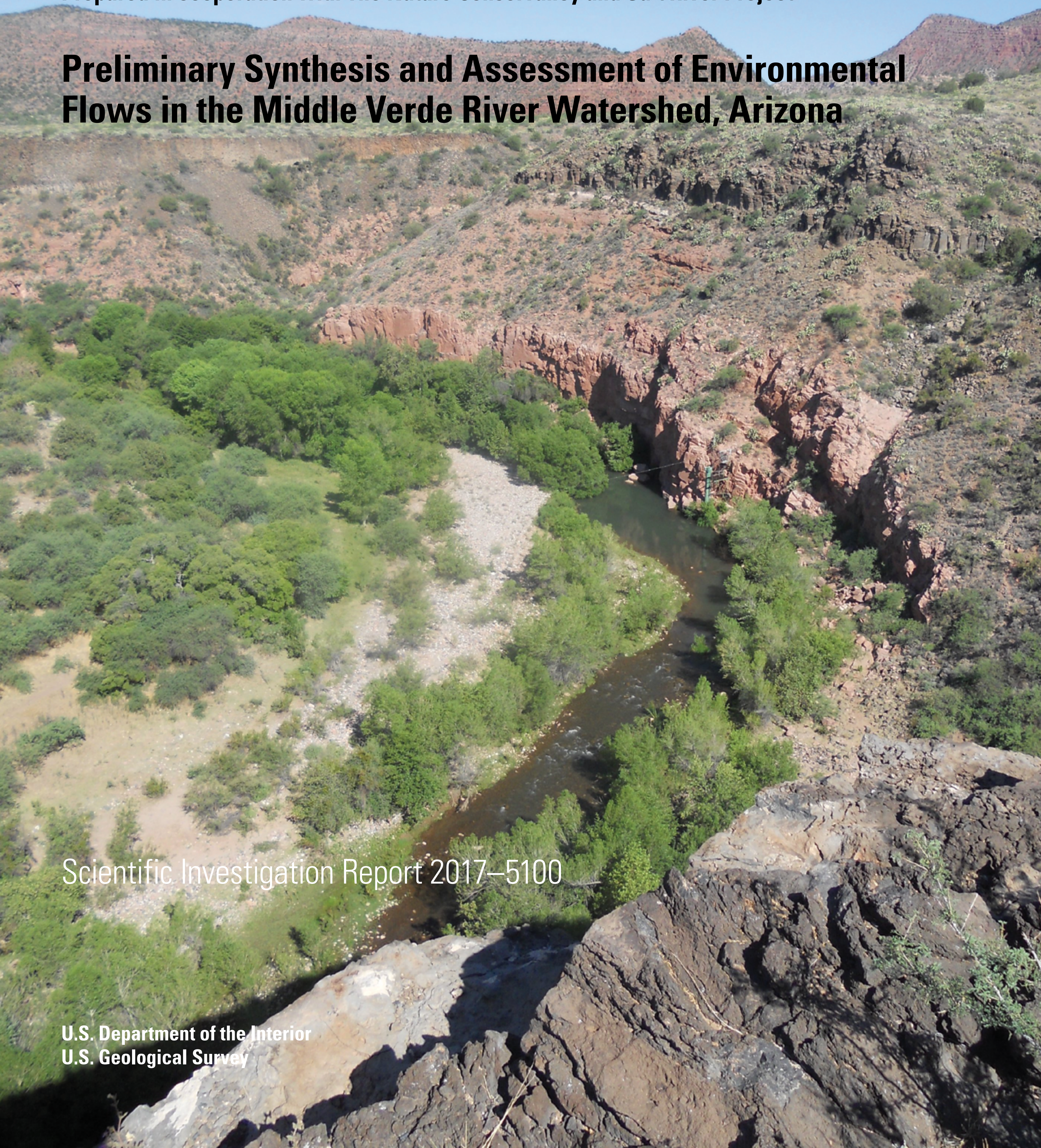
Preliminary Synthesis and Assessment of Environmental Flows in the Middle Verde River Watershed, Arizona



Scientific Investigation Report 2017–5100

U.S. Department of the Interior
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COVER
View looking northwest of the Verde River north of Clarkdale, Arizona. River flow is toward the viewer. U.S. Geological Survey streamgage 09504000 stilling well and cable way are located on the bank. (U.S. Geological Survey photograph by Anne M.D. Brasher.)

BACK COVER
A desert sucker (*Catostomus clarki*) from West Clear Creek, a tributary to the Verde River near Camp Verde, Arizona (U.S. Geological Survey photograph by Nicholas V. Paretti).

Preliminary Synthesis and Assessment of Environmental Flows in the Middle Verde River Watershed, Arizona

By Nicholas V. Paretti, Anne M.D. Brasher, Susanna L. Pearlstein, Dena M. Skow, Bruce Gungle, and Brad D. Garner

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RYAN K. ZINKE, Secretary

U.S. Geological Survey
James F. Reilly II, Director

U.S. Geological Survey, Reston, Virginia: 2018

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Suggested citation:
Paretti, N.V., Brasher, A.M.D., Pearlstein, S.L., Skow, D.M., Gungle, Bruce, and Garner, B.D., 2018, Preliminary synthesis and assessment of environmental flows in the middle Verde River watershed, Arizona: U.S. Geological Survey Scientific Investigations Report 2017–5100, 104 p., https://doi.org/10.3133/sir20175100.

ISSN 2328-0328 (online)

Acknowledgments

This project was developed and funded in cooperation with The Nature Conservancy (TNC). Dan Campbell (retired), Jeanmarie Haney, and Kimberly Schoneck at TNC provided crucial support and expertise regarding the Verde River watershed. Additional funding from the Salt River Project and the U.S. Geological Survey (USGS) WaterSMART program made possible the extension of the project to the development of study sites, sampling, and sample analysis.

Twice during the project, we invited a work group to meet and provide comments and suggestions concerning the direction and focus of our work to keep it relevant to the goals of developing information useful for resource policy and management. Jeanmarie Haney (TNC), Shaula Hedwall (U.S. Fish and Wildlife Service), Charles Paradzick (Salt River Project), John Rasmussen (Yavapai County Water Advisory Committee Coordinator), Patti Spindler (Arizona Department of Environmental Quality, ADEQ), Abe Springer (Northern Arizona University), and David Ward (USGS Grand Canyon Monitoring and Research Center) contributed many insights that strengthened the project.

Data and analyses directly incorporated into the report or used as background information were generously shared by agencies and individuals. Charlie Cave (Yavapai County Flood Control District) provided aerial photographs and detailed topographic maps commissioned by Yavapai County; the hydrological consulting firm HDR, with the permission of Yavapai County, provided a detailed report documenting their hydraulic modeling for the county. Patti Spindler (ADEQ) and Andy Clark (Arizona Game and Fish Department) generously shared valuable macroinvertebrate and fish datasets that were crucial to the report's primary analyses. Jocelyn Gretz provided valuable support for field work and planning. Steve Wiele provided much valuable assistance and support in the development of the project generally and to the section on surface water and groundwater in particular.

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Conversion Factors

U.S. customary units to International System of Units

Multiply	By	To obtain
	Length	
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	4,047	square meter (m^2)
acre	0.4047	hectare (ha)
acre	0.4047	square hectometer (hm^2)
acre	0.004047	square kilometer (km^2)
square foot (ft^2)	0.09290	square meter (m^2)
square mile (mi^2)	259.0	hectare (ha)
square mile (mi^2)	2.590	square kilometer (km^2)
	Flow rate	
acre-foot per year (acre-ft/yr)	1,233	cubic meter per year (m^3/yr)
acre-foot per year (acre-ft/yr)	0.001233	cubic hectometer per year (hm^3/yr)
foot per second (ft/s)	0.3048	meter per second (m/s)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
micrometer (µm)	0.00003937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square hectometer (hm^2)	2.471	acre
square kilometer (km^2)	247.1	acre
square meter (m^2)	10.76	square foot (ft^2)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Flow rate	
cubic meter per year (m^3/yr)	0.000811	acre-foot per year (acre-ft/yr)
cubic hectometer per year (hm^3/yr)	811.03	acre-foot per year (acre-ft/yr)
meter per second (m/s)	3.281	foot per second (ft/s)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F = (1.8 \times °C) + 32.$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$°C = (°F - 32) / 1.8.$$

Datum

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Altitude, as used in this report, refers to distance above the vertical datum.

Supplemental Information

Concentrations of chemical constituents in water are given in grams per liter (g/L).

Abbreviations and Acronyms

ADEQ	Arizona Department of Environmental Quality
AZGFD	Arizona Game and Fish Department
acre-ft/yr	acre-feet per year
BCM	Basin Characteristic Model
BFR	base-flow recession
ft^3/s	cubic feet per second
ft/s	foot per second
DOY	day of year
EFASC	Environmental Flow Allocation and Statistics Calculator
ELOHA	Ecological Limits of Hydrologic Alteration
EMAP	EPA Environmental Monitoring and Assessment Program
EPA	U.S. Environmental Protection Agency
ET	evapotranspiration
EVI	enhanced vegetation index
ft	foot
g/L	gram per liter
GCU	geomorphic channel unit
GIS	geographic information system
GPS	Global Positioning System
HEC-RAS	Hydrologic Engineering Center River Analysis System
km	kilometer
LOWESS	locally weighted scatterplot smoothing
m	meter
mm	millimeter
MODFLOW	USGS Modular Finite-Difference Flow Model
MODIS	Moderate Resolution Imaging Spectroradiometer
NARGFM	Northern Arizona Groundwater Flow Model
NAU	Northern Arizona University
NAWQA	USGS National Water Quality Assessment program
NMDS	nonmetric multidimensional scaling
NRMF	normalized range of monthly flow
NWIS	USGS National Water Information System
PrAMA	Prescott Active Management Area
PRISM	Parameter-Elevation Regressions on Independent Slopes Model
TNC	The Nature Conservancy
USFS	U.S. Forest Service
USGS	U.S. Geological Survey
τ	Kendall's tau correlation coefficient

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Abstract

A 3-year study was undertaken to evaluate the suitability of the available modeling tools for characterizing environmental flows in the middle Verde River watershed of central Arizona, describe riparian vegetation throughout the watershed, and estimate sediment mobilization in the river. Existing data on fish and macroinvertebrates were analyzed in relation to basin characteristics, flow regimes, and microhabitat, and a pilot study was conducted that sampled fish and macroinvertebrates and the microhabitats in which they were found. The sampling for the pilot study took place at five different locations in the middle Verde River watershed. This report presents the results of this 3-year study.

The Northern Arizona Groundwater Flow Model (NARGFM) was found to be capable of predicting long-term changes caused by alteration of regional recharge (such as may result from climate variability) and groundwater pumping in gaining, losing, and dry reaches of the major streams in the middle Verde River watershed. Over the period 1910 to 2006, the model simulated an increase in dry reaches, a small increase in reaches losing discharge to the groundwater aquifer, and a concurrent decrease in reaches gaining discharge from groundwater. Although evaluations of the suitability of using the NARGFM and Basin Characteristic Model to characterize various streamflow intervals showed that smaller-scale basin monthly runoff could be estimated adequately at locations of interest, monthly stream-flow estimates were found unsatisfactory for determining environmental flows.

Orthoimagery and Moderate Resolution Imaging Spectroradiometer data were used to quantify stream and riparian vegetation properties related to biotic habitat. The relative abundance of riparian vegetation varied along the main channel of the Verde River. As would be expected, more upland plant species and fewer lowland species were found in the upper-middle section compared to the lower-middle section, and vice-versa. Vegetation changes within the upper-middle and lower-middle reaches are related to differences in climate and hydrology. In general, the riparian vegetation of the middle Verde River watershed is that of a healthy ecosystem's mixed age, mixed patch structure, likely a result of the mostly unaltered disturbance regime.

The frequency of in-river hydrogeomorphic features (pool, riffle, run) varied along the middle Verde River channel. There was a greater abundance of riffle habitat in the upper-middle reach; the lower-middle reach included more pool habitat. The Oak Creek tributary was more homogenous in geomorphic stream habitat composition than West Clear Creek, where runs dominated the upper reaches and pools dominated many of the lower reaches.

On the basis of the period of record and discharges recorded at 15-minute intervals, five flows were found to reach the gravel-transport threshold. Sediment mobilization computed with flows averaged over daily time steps yielded just three flows that reached the gravel-transport threshold, and monthly averaged flows yielded none. In the middle Verde River watershed, 15-minute data should be used when possible to evaluate sediment transport in the river system.

Data from more than 300 fish surveys conducted from 1992 to 2011 were analyzed using two schemes, one that divided the river into five reaches based on basin characteristics, and a second that divided the river into five reaches based on degree of flow alteration (specifically, diversions). Fish community metrics and assemblage data were used to analyze patterns of species composition and abundance in the two approaches. Overall, native and non-native species were regularly interacting and probably competing for similar resources. Fish abundances were also analyzed in response to floods and other flow metrics. Although the data are limited, native fish abundances increased more rapidly than non-native fish abundances in response to large floods. The basin-characteristic reach analysis showed native fish in greater abundance in the upper-middle reaches of the Verde River watershed and generally decreasing with downstream distance. The median relative abundance of native fish decreased by 50 percent from reach 1 to reach 5. Using the reach scheme based on degree of flow alteration, nondiverted reaches were found to have a greater abundance of native fish than diverted reaches. In heavily diverted reaches, non-native species outnumbered native species.

Fish metrics and stream-flow metrics for the 30, 90, and 365-day periods before collection were computed and the results analyzed statistically. Only abundance of all fish species was associated with the 30-day flow metrics. The 90-day

flow metrics were generally positively associated with fish metrics, whereas the 365-day flow metrics had more negative correlations. In particular, significant relations were found between fish metrics and the magnitude and frequency of high flows, including maximum monthly flow, median annual number of high-flow events, and median annual maximum streamflow. Native sucker (Catostomidae) populations tended to decrease in periods of extended base flow, and fish in the non-native sunfish family (Centrarchidae) decreased in periods of flashy, high magnitude flows.

A pilot study surveyed fish at five locations in the upper part of the middle Verde River watershed as a means to measure microhabitat availability and quantify native and non-native fish use of that available microhabitat. Results indicated that native and non-native species exhibit some clear differences in microhabitat use. Although at least some native and non-native fish were found in each velocity, depth, and substrate category, preferential microhabitat use was common. On a percentage basis, non-native species had a strong preference for slow-moving and deeper water with silt and sand substrate, with a secondary preference for faster moving and very shallow water and a coarse gravel substrate. Native species showed a general preference for somewhat faster, moderate depth water over coarse gravel and had no clear secondary preference.

Macroinvertebrate-variables index period, high-flow year, and collection location (upper-middle Verde River, lower-middle Verde River, or Verde River tributaries) were found to be important explanatory variables in differentiating among community metrics. Overall richness (number of unique taxa), Shannon's diversity index, and the percent of the most dominant taxa were all highly correlated, but their response to each macroinvertebrate variable was different. The percentage of mayfly (order Ephemeroptera) taxa was significantly higher in Oak Creek and the upper-middle and lower-middle Verde River reaches, locations which have higher flows and more urbanization than other reaches. When community metrics were related to hydrologic metrics, caddisfly (order Trichoptera) populations appeared to increase and mayfly populations to decrease in response to less flashy and more stable streamflows. Conversely, caddisfly populations appeared to decrease and mayfly populations to increase in response to greater flow variability.

Six locations along the Verde River were sampled for macroinvertebrates as part of a pilot study associated with this report—(1) below Granite Creek, (2) near Campbell Ranch, (3) at the U.S. Geological Survey Paulden gage, (4) at the Perkinsville Bridge, (5) at the USGS Clarkdale gage, and (6) near the Reitz Ranch property. A nonmetric multidimensional scaling ordination of macroinvertebrate assemblages showed that the Verde River below Granite Creek site was different from the five other sites and that the Perkinsville Bridge and near Reitz Ranch samples had similar community structure. The near Campbell Ranch and Paulden gage locations had similar microhabitat characteristics, with the exception of riparian cover, yet the assemblage structure was very different. The different community composition at Verde River below Granite Creek was likely due to it having the smallest substrate sizes, lowest velocities, shallowest depths, and most riparian cover of the six sites.

Introduction

The Verde River (fig. 1) is among the largest streams in Arizona, providing about 40 percent of the surface water delivered to the Phoenix metropolitan area by the Salt River Project (The Nature Conservancy, 2009). Its watershed covers 4.2 million acres in central Arizona, and includes about 500 miles of perennial streams, including the Verde River, and is the only watershed in the State with designated "wild and scenic" reaches as described in the Wild and Scenic Rivers Act (1968), Public Law 90-542; 16 U.S.C. 1271 et seq. (National Wild and Scenic Rivers System, 2016a; National Wild and Scenic Rivers System, 2016b). It was listed in the 2006 edition of America's Most Endangered Rivers (American Rivers, 2006) based on the apparent threat groundwater pumping may pose to Verde River base flows. Historically, at least 13 native fish species lived in the watershed, including seven that are now threatened or endangered (The Nature Conservancy, 2009). The Verde River watershed supports one-third of the breeding areas for the desert-nesting bald eagle (*Haliaeetus leucocephalus*), populations of the southwestern willow flycatcher (*Empidonax traillii extimus*) and yellow-billed cuckoo (*Coccyzus americanus*), and more than 200 other bird species that use the river's riparian habitat (The Nature Conservancy, 2009). The flows that occur in the middle Verde River watershed are largely unregulated and are less affected by urban use, urban and agricultural runoff, and channelization than in the lower Verde River watershed.

The Verde River is central to the way of life for the many towns and communities located in the watershed and is part of the ancestral home for the Yavapai-Apache Nation, the Fort McDowell Yavapai Nation, and other tribes (The Nature Conservancy, 2009). Although large parts of the Verde River watershed remain undeveloped and are administered by the U.S. Forest Service (USFS), there is considerable commercial and residential development occurring in the Prescott area and in the Verde Valley (Blasch and others, 2006). The related population growth will likely result in an increased demand on the region's water resources, and there is concern about how these growing human water needs will continue to be met in the upper and middle Verde River watersheds while preserving its important ecological values (Haney and others, 2008).

For example, the City of Prescott has statutory rights to pump up to 14,000 acre-ft of groundwater annually from Big Chino Valley, although the Director of the Arizona Department of Water Resources (ADWR) issued a determination in 2008 that recognized 8,067.4 acre-ft as the annual volume to which the City of Prescott is entitled (Arizona Department of Water Resources, 2014). Additional groundwater is legally available to municipalities in the Prescott AMA as retired historically irrigated acres (Arizona Department of Water

Resources, 2014). Prescott currently plans to collaborate with Prescott Valley to pump water from the Big Chino Valley and transport it to their respective municipalities (Marder, 2009; Auer, 2010). The Salt River Project, which is the main water utility for the metropolitan Phoenix area, has claims to Verde River surface water that predate Arizona statehood (Marder, 2009). Garner and others (2013) found that base flow at the Clarkdale gaging station has declined by about 4,900 acre-ft/yr as a result of human stresses placed on the flow system. For the future, model simulations show a decrease in Verde River base flow (Garner and others, 2013; Jaeger and others, 2014) as well as decreased stream connectivity (Jaeger and others, 2014). This is the result of both climate forecasts that project increased aridity in the southwest (Williams and others, 2010) and increased human stress from pumping and diversions (Garner and others, 2013).

Environmental Flows

"Environmental flows" is a term that has gained currency over the past 15 years. It has evolved from a focus on minimum flows required for species preservation to one that recognizes the importance of natural flow regimes. It is based on the concept that the natural dynamic character of the flow regime of a river or stream is necessary to sustain riverine ecosystems intact. Streams altered by human modifications, such as dams, diversions, or channel engineering, can also be managed to mimic an environmental flow regime that meets ecological and social objectives (Acreman and others, 2014). The flow characteristics necessary to sustain riverine ecosystems, specifically species' habitats and thus health and diversity, are thus referred to as the river's "environmental flows."

To understand the consequences to the ecosystem of various water-use scenarios, it is necessary to understand the system's environmental flows—the interactions between hydrology and ecology that form the basis for the water needs of the ecosystem. Ecosystem water needs include, but are not limited to, consumptive use by riparian vegetation and habitat requirements of in-stream biota (fish and macroinvertebrates). The hydrologic characteristics that affect ecosystem water needs include both streamflow regime (magnitude, frequency, timing, and variability of flow) and groundwater conditions (such as depth to groundwater and annual groundwater fluctuations) (Poff and others, 2010; Konrad and others 2008a).

The first step in establishing environmental flows is to develop scientifically credible estimates of flow regimes that sustain properly functioning ecosystems. The natural hydrologic regime of a river or stream can be divided into four components—floods, high flow pulses, base flows, and extreme low flows. Each of these flow components has different ecological functions, and thus all are likely to be important for ecosystem integrity (fig. 2). To represent these flow components of the natural flow regime of any stream system, measures of magnitude, timing, duration, frequency, and rate of change must be developed (Poff and others, 1997). The magnitude of an event refers to the volume of water passing a given point over a period of time. Timing is the predictability or seasonality of a given magnitude. The duration is the amount of time a given magnitude persists. Frequency refers to how often a magnitude occurs over a specific period of time, and the rate of change is how quickly a flow goes from one state to another. Hydrologic alteration can refer to changes in any of these components.

Southwestern desert streams, including those of the Verde River watershed, are characterized by large variations in flow magnitude, including large floods originating from high-intensity summer thunderstorms and extended winter frontal-precipitation events (fig. 2; Poff and others, 1997; Blasch and others, 2006). However, the Verde River and its tributaries are most commonly in a base-flow regime—streamflow is supported entirely by groundwater discharge to the stream channel, including discharge from numerous springs. The spatial connectivity of flowing reaches in the Verde River also vary seasonally, with more connectivity in the winter and less in the late spring and early summer, and this has specific implications for aquatic species (Jaeger and others, 2014). This makes the flow regimes of semiarid southwestern streams like the Verde River highly dependent on groundwater/surface-water interactions, and as a result, groundwater conditions such as depth to groundwater and annual groundwater level fluctuations need to be incorporated into any analysis of streamflow.

ELOHA Framework

Flow regime is a primary determinant of the structure and function of aquatic and riparian ecosystems. The Ecological Limits of Hydrologic Alteration (ELOHA) is a framework that incorporates scientific information on hydrology and biotic-habitat requirements with a decision-making process based on societal values and management needs (Richter and others, 2003; Richter and others, 2006; Poff and others, 2010). Part of this process includes quantifying flow-ecology relations that can be used as the basis for determining environmental flows.

The ELOHA framework is based on a spatially comprehensive hydrologic foundation and generalized flow-ecology relations that can be applied across a region to establish environmental flow standards. The purpose of environmental-flow standards is to protect entire ecosystems and the range of conditions required for the success of aquatic communities rather than maintenance of a minimum flow threshold to sustain individual species. Once the hydrologic foundation has been defined (fig. 3, step 1) and the river or stream classified (fig. 3, step 2), the ecological responses to hydrologic (flow) alteration can be defined (fig. 3, step 3). Then explicit societal decisions about the level of protection (which can vary by river or stream) can be made and administrative mechanisms (water permits, land-use planning, water trusts) implemented for flow standards (fig. 3, social process) (Poff and others, 2010). Various studies have addressed steps 1 and 2 (for example, Wirt, 2004; Wirt and others, 2005; Blasch and others, 2006; Springer and Haney, 2008; Garner and Bills, 2012).

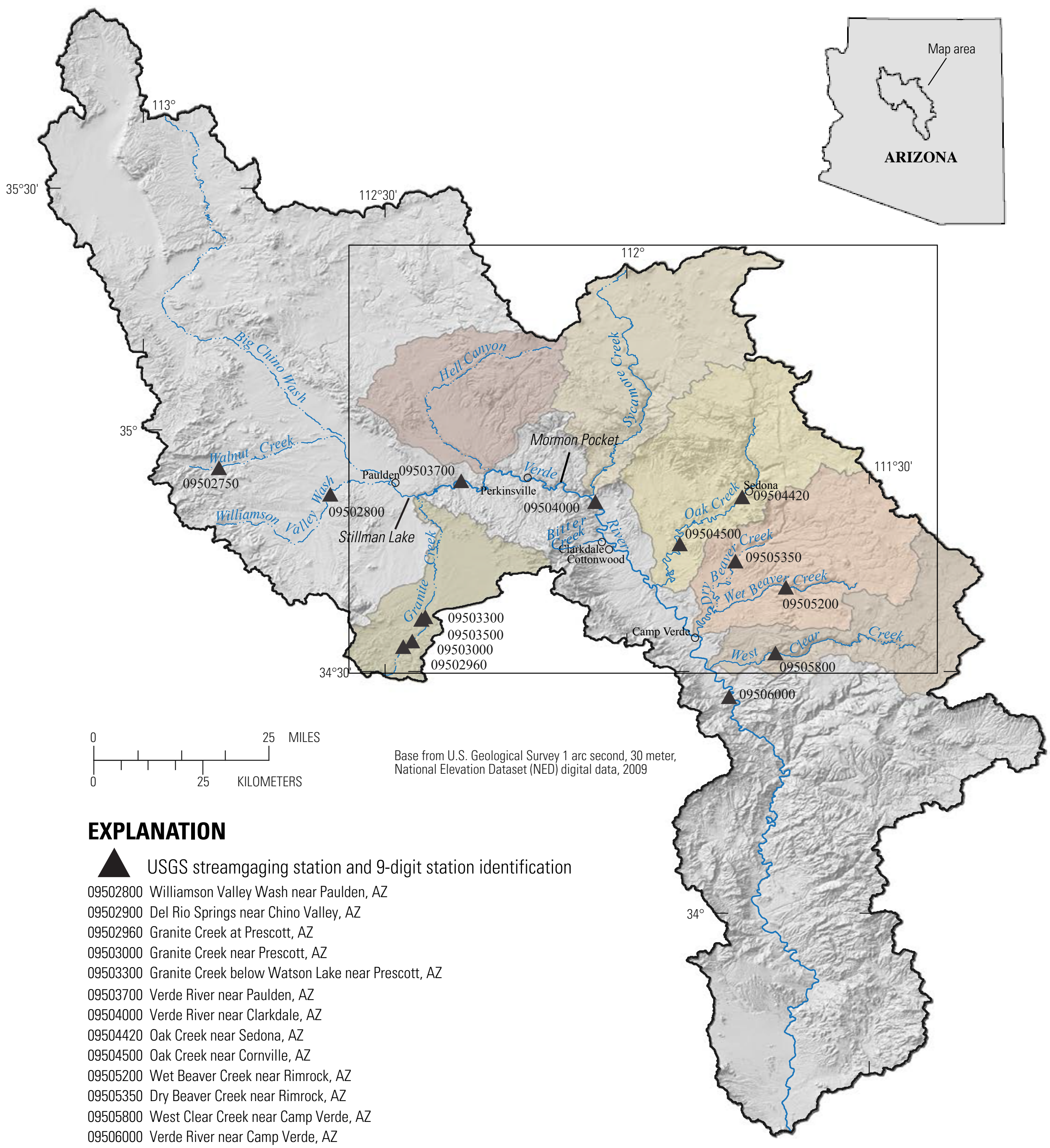


Figure 1. Map of the Verde River watershed, Arizona, including streamgaging stations, major towns, and tributary subwatersheds that are referred to in this report.

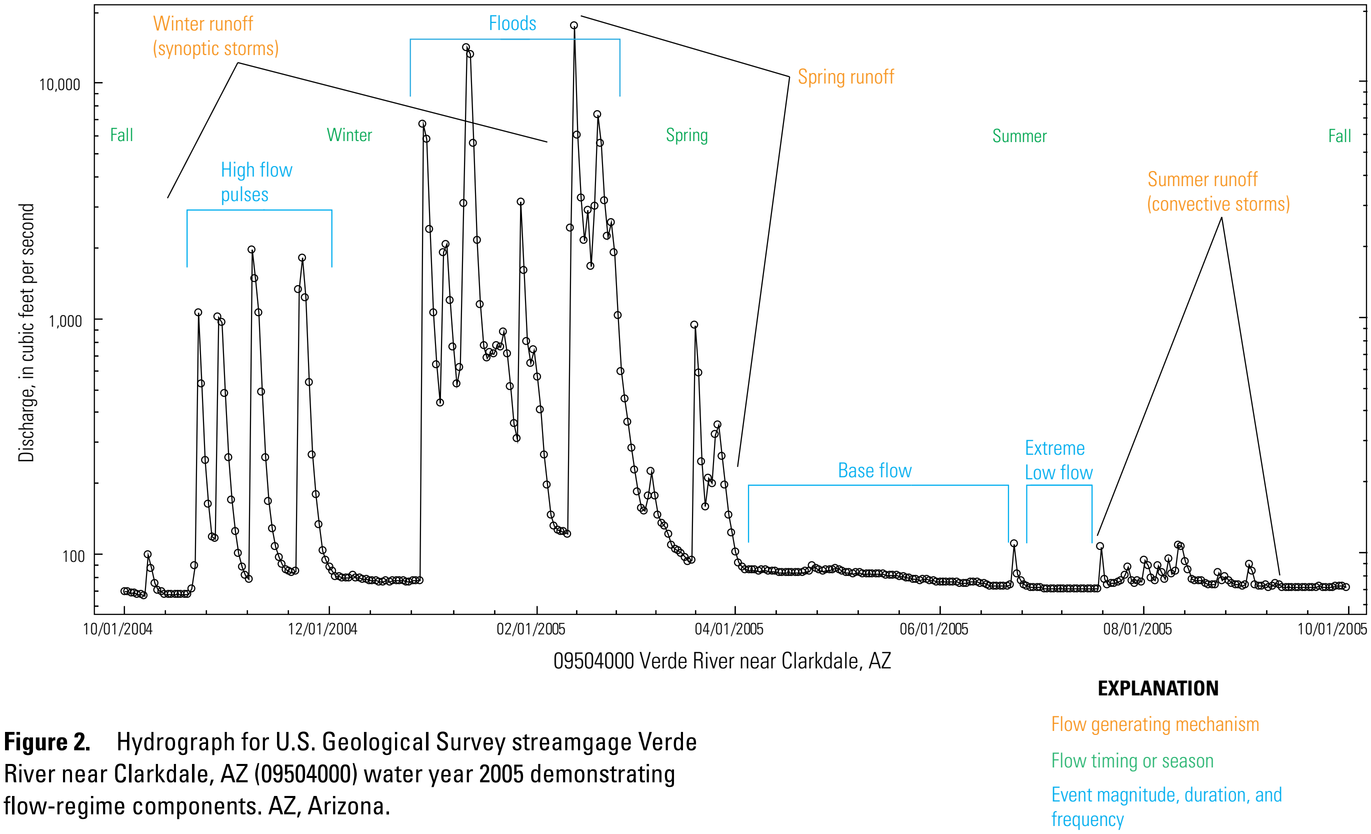


Figure 2. Hydrograph for U.S. Geological Survey streamgage Verde River near Clarkdale, AZ (09504000) water year 2005 demonstrating flow-regime components. AZ, Arizona.

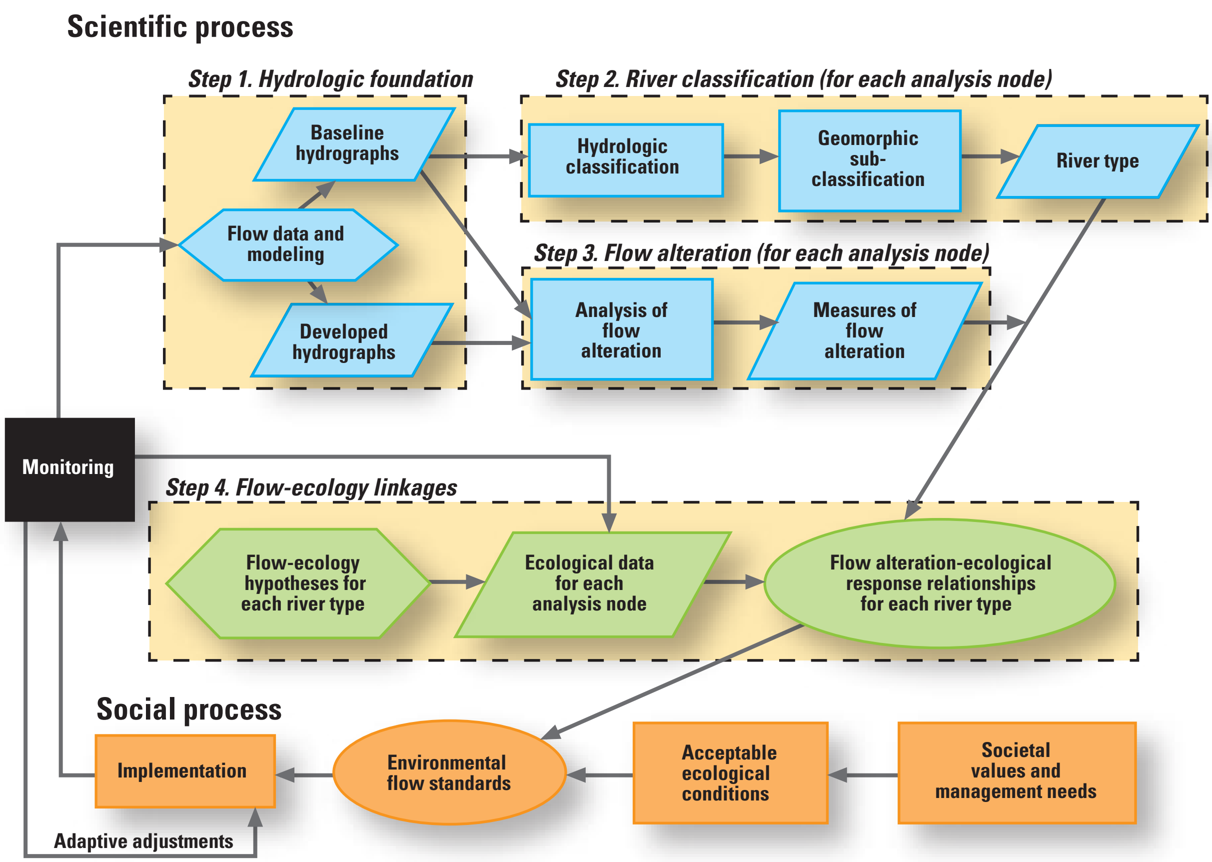


Figure 3. Diagram showing a conceptual model of environmental flows (from Poff and others, 2010).

The present study addresses step 3, the scientific process of defining ecological responses to hydrologic alteration.

Purpose and Scope

The purpose of this report was to compile existing hydrological and ecological data to better understand environmental flows in the setting of the middle Verde River watershed. To accomplish this, the connection between abiotic and biotic ecosystem drivers and how each relates to the factors that make up environmental flows in the Verde River was investigated. In addition, hydrologic regime and the associations of hydrologic characteristics with biotic communities were quantified. This report summarizes a 3-year project investigating the connection between abiotic and biotic ecosystem drivers and how each relates to the factors that make up environmental flows in the Verde River (fig. 4). It also quantifies hydrologic regime and the associations of hydrologic characteristics with biotic communities. This project consists largely of obtaining and summarizing existing biological and hydrological data, as well as conducting initial analyses and pilot studies on riparian vegetation characteristics, microhabitat use by fish, and macroinvertebrate-community structure. Geomorphic features related to habitat were mapped and quantified, and mobilization of bed sediments was examined at sites where sufficient data were available. Additional related components were the evaluation of the Northern Arizona Regional Groundwater Flow Model (NARGFM) as an aid in ecological studies and the estimation of monthly streamflow budgets in the Verde River and its major tributaries.

Our objectives were to improve understanding of flow ecology (hydrology and ecological associations) in semiarid watersheds and to develop and quantify associations between hydrologic alteration and ecological responses to provide a scientific foundation for ecologically sustainable water management. The study was intended to serve as a resource to other agencies for developing techniques and methods widely applicable across the Southwest.

Physical Setting

The Verde River is located in Central Arizona and flows for 189 miles from below Sullivan Dam, near the town of Paulden, past the towns of Cottonwood and Camp Verde, eventually passing through Horseshoe and Bartlett Reservoir, and ultimately to its confluence with the Salt River near Fort McDowell (The Nature Conservancy, 2009; Garner and others, 2013; fig. 1). The headwaters begin in the region of largely ephemeral washes in the Prescott Valley and Chino Valley area and in the Big Chino Wash drainage. Perennial flow begins in the upper Verde River just below Sullivan Lake with additions to base flow coming from springs near the confluence

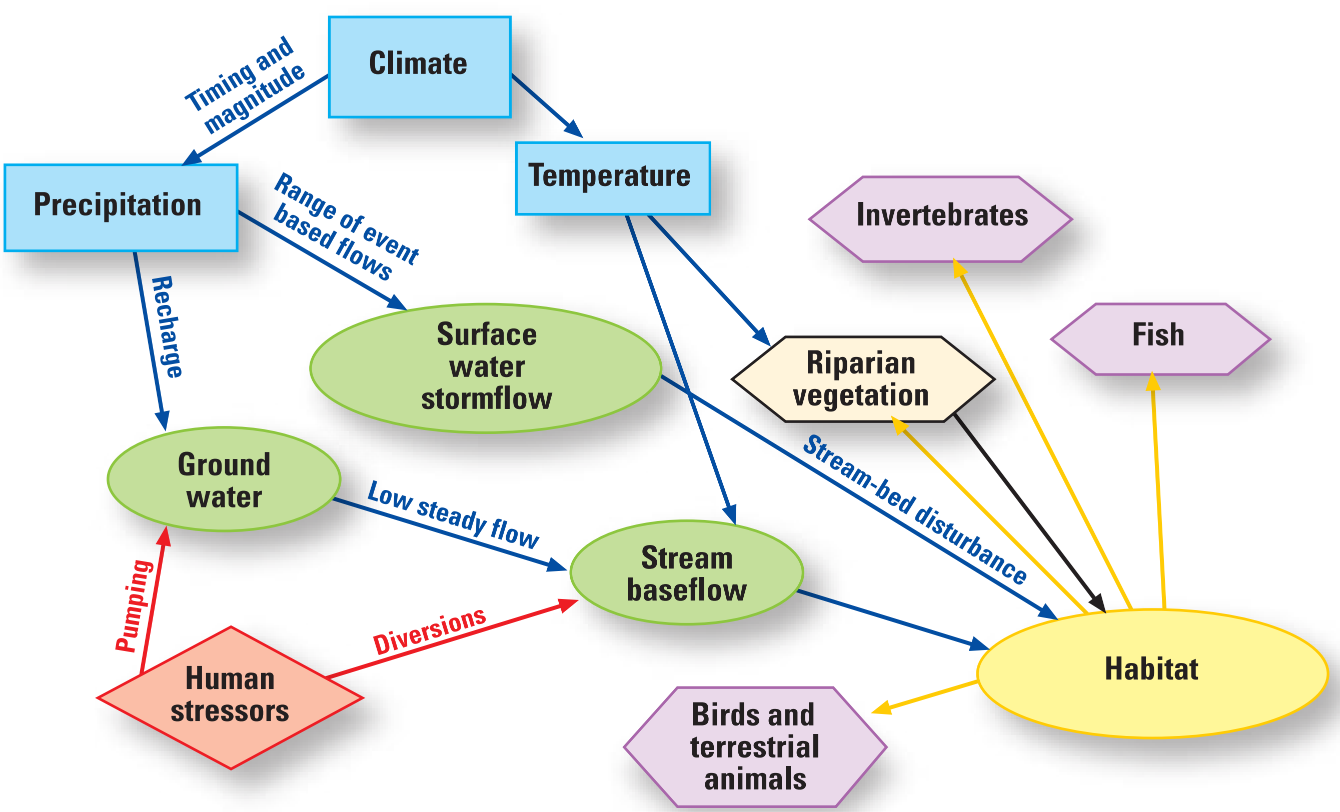


Figure 4. Diagram showing a conceptual model of abiotic and biotic ecosystem drivers.

with Granite Creek, about 2 miles downstream of the town of Paulden (Springer and Haney, 2008). Major intermittent and ephemeral tributaries in this section of the river include Big Chino Wash, Williamson Valley Wash, Walnut Creek, Granite Creek, and Hell Canyon. The lower-middle Verde River begins at the U.S. Geological Survey (USGS) Clarkdale gaging station (09504000) (Blasch and others, 2006). Major tributaries in this section are Sycamore Creek, Oak Creek, Beaver Creek, and West Clear Creek. Perennial flow for the upper 26 miles of the Verde River is largely fed by inflow from the Big Chino and Little Chino aquifers (Wirt and others, 2005), which underlie grasslands to the north, west, and south. Additional groundwater enters the mainstem near Perkinsville and Mormon Pocket from consolidated rock aquifers on the southern Colorado Plateau. Streamflow in the lower-middle Verde River is sustained by surface runoff, groundwater discharge from the basin-fill aquifer in the Verde Valley, and contributions from the tributaries in the lower-middle Verde River watershed (Springer and Haney, 2008). In the Verde Valley, the river's major tributaries are largely fed by springs that drain the C and Redwall-Muav aquifers, which collect water above the Mogollon Rim (Springer and Haney, 2008).

The upper and middle Verde River watersheds have a semiarid climate. Precipitation in central Arizona tends to be strongly seasonal, dominated by low-intensity, long-duration winter storms and shorter, high-intensity local summer storms during the North American monsoon (Adams and Comrie, 1997; Blasch and others, 2006). As drainage area increases, relative magnitude of streamflow generated by intense convective storms decreases (House and Hirschboeck, 1997). Winter storms produce the most and largest floods because of stronger orographic effects and more frequent passing of frontal storms (House and Hirschboeck, 1997), whereas snowmelt in the spring can provide the river with steady high flows. An abundance of microclimates exists due to elevation, slope, and canyon width and depth, which all affect sun exposure in a particular area (Blasch and others, 2006). Detailed and comprehensive discussions of Verde River hydrology and its ecological associations have been presented by Springer and Haney (2008) and Blasch and others (2006).

Surface Water and Groundwater

In establishing environmental flows, all features of the stream hydrograph are of interest. Biota are affected by and adapted to the full range and timing of flows. Flows can originate from different mechanisms such as synoptic, tropical, or convective precipitation events. Discharge changes can also be generated by seasonal shifts that will result in snowmelt runoff when temperatures increase. In periods of high temperatures and low precipitation, groundwater contributions will sustain base-flow conditions in the stream channel.

Metrics of flow frequency, rate of change, magnitude, and duration can be used in classifying environmental flows (fig. 2). The USGS operates multiple stream-gaging stations on the upper and middle Verde River and its tributaries from Sullivan Dam to below Camp Verde (table 1). The Salt River Project operates a low-flow gage at Campbell Ranch near the Verde River perennial headwaters. The Yavapai County Flood Control District operates flood warning sensors throughout the upper and middle Verde River watersheds, including one at Perkinsville. Discharge in lower Granite Creek is heavily affected by reservoirs near Prescott, and the discharge in the Verde River below Camp Verde is affected by diversions in Cottonwood. Diversions upstream from the Oak Creek near Cornville stream-gaging station (09504500) affect the discharge record there as well.

Surface-Water Hydraulic Models

In-channel stream habitat is closely linked to flow properties. Hydraulic models can be used to predict how those flow properties vary with discharge to help assess how in-stream habitat is affected by a range of discharges. One-dimensional

Table 1. U.S. Geological Survey mainstem and tributary streamflow gaging stations in the middle Verde River watershed, Arizona.

Streamgage name	Station number	Years of record	Mean streamflow over period of record, in cubic feet per second
Verde River near Paulden, AZ	09503700	48	43
Verde River near Clarkdale, AZ	09504000	49	178
Verde River near Camp Verde, AZ	09506000	34	400
Granite Creek at Prescott, AZ	09502960	16	6
Granite Creek near Prescott, AZ	09503000	32	7
Granite Creek below Watson lake near Prescott, AZ	09503300	12	4
Oak Creek near Sedona, AZ	09504420	29	82
Oak Creek near Cornville, AZ	09504500	67	87
Dry Beaver Creek near Rimrock, AZ	09505350	50	42
Wet Beaver Creek near Rimrock, AZ	09505200	45	32
West Clear Creek near Camp Verde, AZ	09505800	45	61

hydraulic models have been completed for reaches of the Verde River in Clarkdale and Cottonwood. Ross and others (2010) constructed a flow model as part of a study of diversions of Verde River water in that area. The model computes steady, cross-sectionally averaged stage and velocity as functions of specified discharge. The model was used by Ross and others (2010) to examine the effects of stream-flow diversions in Verde Valley and included monitoring of water-surface levels near diversions.

The Yavapai County Flood Control District commissioned HDR, Inc., to construct a one-dimensional model for the reach of the Verde River between the Clarkdale and Camp Verde gages, the reach that includes the cities of Clarkdale, Cottonwood, and Camp Verde (HDR, Inc., 2011). This model was designed and implemented for delineation of flood-inundation levels. In addition to developing the model, HDR, Inc. (2011), developed stream-flow statistics for the Verde River to supply discharges at specified recurrence intervals.

Groundwater Models

Base flow is a crucial component of environmental flows, especially in a watershed with ephemeral and intermittent reaches, such as in the Verde River watershed. Groundwater discharge to stream courses is the source of base flow, and reductions in groundwater discharge will reduce base flow and can expand the length of dry reaches and increase persistence of dry reaches over time. Groundwater-flow models can be used to study the effects groundwater withdrawals or reductions in recharge (such as could result from changes in climate) might have on base flow.

The Arizona Department of Water Resources (ADWR) developed a groundwater model of the Prescott Active Management Area (PrAMA; Corkhill and Mason, 1995) which has since been updated (Nelson, 2002; Timmons and Springer, 2006). The model domain is bounded by the PrAMA, and does not extend into the perennial section of the Verde River near the confluence with Granite Creek. The model has been used to study predevelopment conditions and the effects of groundwater pumping on groundwater levels within the PrAMA.

Northern Arizona Groundwater Flow Model

The Northern Arizona Groundwater Flow Model (NARGFM; Pool and others, 2011) was developed by the USGS in response to growing concerns about groundwater resources in the northern part of the State. Pool and others (2011) list two primary uses for the model—(1) identification of data needs and (2) prediction of changes in groundwater levels and groundwater discharge to streams, springs, and riparian vegetation as a result of groundwater use and management practices. The construction of a numerical model of a complicated system provides a rigorous framework for assembling information about the physical system. The groundwater model can also be used to predict system responses to changes in recharge and groundwater withdrawals over time.

Model Construction

NARGFM extends from Kane County in southern Utah to the north, from just east of the California border to the west, east into McKinley, Cibola, and Catron County of western New Mexico, and south to the Verde River and Salt River watersheds (fig. 5). The construction of NARGFM was preceded by a study of the upper and middle Verde River watershed hydrology (Blasch and others, 2006) in which an extensive dataset was compiled that included surface-water discharges, groundwater-well levels, precipitation, geochemistry, and geological features. NARGFM predicts groundwater contributions to streams but does not include surface runoff into streams. Consequently, only changes to base flow can be represented by the model; flows that are a direct result of snowmelt or precipitation runoff must be considered independently. Major streams are represented in the model, including the Verde River and its major tributaries.

The complex geology in northern Arizona is represented by three layers in the model. Parameters (hydraulic conductivity, specific storage, and specific yield) and the topography and layer thickness vary over the model domain to represent variations in hydraulic properties. Model parameters were calibrated primarily with well water levels and streamflow data. The calibration process and results are extensively described by Pool and others (2011), with comparisons between model predictions and data. Areas where information is sparse become especially apparent in the calibration process if information is insufficient to define parameter values to within reasonable ranges. Pool and others (2011) describe in detail where and what types of new data would improve the model calibration. Model limitations are also discussed in detail.

The model has both steady-state and transient applications. The steady-state version has no time dependence and computes an equilibrium condition; this application was used to retrodict predevelopment conditions in 1910. The transient applications (model predictions vary in time in response to varying recharge and pumping stresses) were applied from 1910 to 2006.

Use of NARGFM for Environmental Flow Studies

Spatial Resolution

NARGFM-predicted near-channel water levels could be a guide for general response of near-stream water levels to varying stresses but on a scale appropriate for the model (fig. 6). The model cannot represent steep gradients nor variations in near-stream water levels on a scale smaller than the 1-kilometer (km) grid due to purely numerical considerations (fig. 6). In other words, NARGFM would be useful for evaluating kilometer (or greater) scale trends but not for subkilometer scale trends. In addition, Quaternary alluvium is not included in the model, and stream-aquifer properties are represented in the model on an even coarser spacing than the grid spacing. Consequently, the model can reasonably predict near-stream

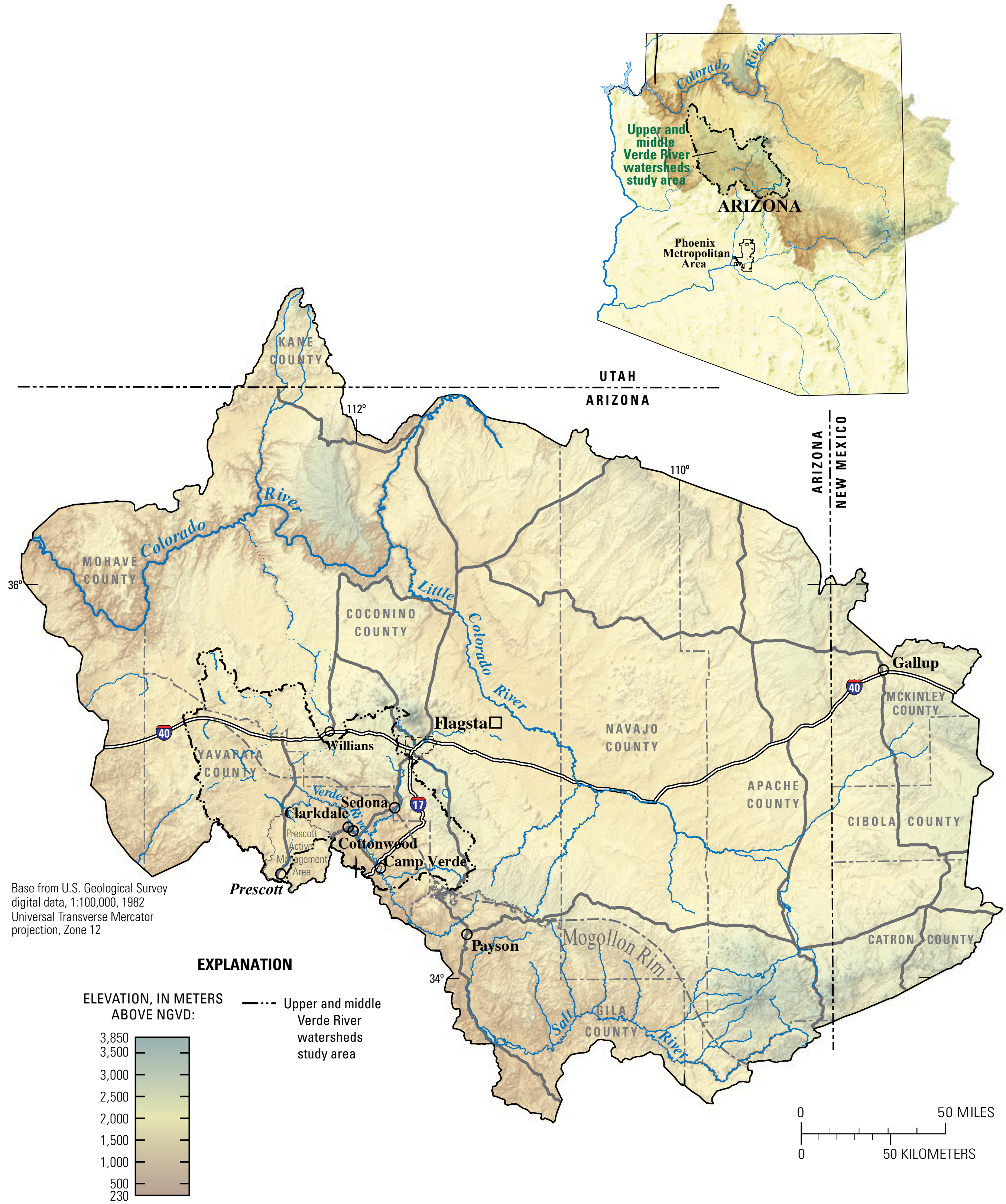


Figure 5. Map of northern Arizona showing area covered by the Northern Arizona Regional Groundwater Flow Model.

Figure 6. Aerial photograph showing the length of an example sampling site near Reitz Ranch on the Verde River, Arizona, relative to the Northern Arizona Regional Groundwater Flow Model grid spacing. The red pixels are sample survey points, whereas the white lines, which are 1 kilometer apart, show the computational grid used by the model.

water levels over a minimum scale of no less than several kilometers.

Temporal Resolution

When operating in transient mode (time varying), the USGS Modular Finite-Difference Flow Model (MODFLOW) allows the user to specify both stress periods and time steps. A stress period is a time during which a set of boundary conditions, or model inputs such as recharge or well pumping, are held constant. A new set of boundary conditions is typically specified in succeeding stress periods. A time step is a time-dependent solution over a time equal to or less than the stress period. Having time steps smaller than the stress period allows the model to better represent the time derivatives in the governing equations with its numerical approximation.

The transient NARGFM starts with conditions that occurred in 1910 followed by a transient period extending from 1910 to 1938. Subsequent stress periods are 10 years, except for the last, which is 6 years long. The stress periods are further divided into five time steps of varying length. The model does not represent any seasonal variations and only represents groundwater contributions to streamflow.

NARGFM Stream-Flow Predictions

Because NARGFM predicts groundwater contributions to streamflow, base flow can be estimated from model results. However, some of the hydrograph components that are important for ecological considerations are not represented by the NARGFM model. Variations in streamflow that results from storms must be considered independently of NARGFM.

The alluvial aquifer, which generally consists of more permeable loosely consolidated fluvial sediments, is not represented in the model, either. Consequently, the interactions between streamflow and groundwater occur on a larger scale consistent with the hydraulic properties of the aquifer; modifications to local streamflow that are a result of groundwater flow through alluvial sediments are not part of the model.

The model predicts stream reaches on the scale of kilometers that gain flow from groundwater, lose flow to groundwater, or are dry (table 2; fig. 7). The interaction between groundwater and streamflow predicted by the model are the average conditions during the model time step. The model predicts some increase in losing and dry reaches between the end of the initial steady-state time period ending in 1910 (fig. 7*A*) and the end of the transient time period ending in 2006 (fig. 7*B*). For example, the model predicts dry reaches in Wet Beaver Creek in 2006 that were generally perennial in 1910.

Estimates of Monthly Streamflow at Ungaged Locations

Streamgages are located on the middle and upper Verde River and some of its tributaries, but other tributaries and subbasins are ungaged (fig. 1). Consequently, tools to estimate streamflow at ungaged sites can be useful for filling in spatial gaps in discharge records at study sites. In this study, we used results from two existing models, the Basin Characteristic Model (BCM; Flint and Flint, 2007) and NARGFM (Pool and others, 2011) to estimate monthly streamflow at ungaged subbasins in the Verde River watershed.

Streamflow can be divided into two components—base flow and storm-generated runoff. Base flow can vary over time in response to variations in large-scale natural or artificial recharge or in response to groundwater pumping. Base flow can vary spatially as a result of the accumulation of groundwater inflow along a channel, variability in geologic strata adjacent to a river, and local variations in the shape and volume of an alluvial aquifer. Storm-generated runoff results from the accumulation of overland flow during storms and generates streamflow responses on a smaller time scale than base-flow responses to regional groundwater flows.

The BCM provides distributed estimates of potential evapotranspiration, soil moisture storage, and in-place recharge and in-place runoff (Flint and Flint, 2007). “In place” refers to model predictions of infiltration and runoff in a cell; flow is not routed by the model over the land surface or into the subsurface. Model predictions are based on vegetation, topography, geology, soil properties, and precipitation and temperature estimates. The precipitation and temperature inputs to the BCM are derived from Parameter-Elevation Regressions on Independent Slopes Model (PRISM) estimates (Daly and others, 1994). The BCM operates on a 270-km grid and a monthly time step. The BCM predictions of in-place infiltration have been used to provide recharge inputs for groundwater modeling and for initial recharge estimates that were modified during the calibration process of NARGFM (Pool and others, 2011). Flint and others (2012) developed a water budget for the San Diego River basin that incorporated BCM estimates of both recharge and runoff, and the runoff component was used to estimate monthly streamflow at ungaged sites. In this study, BCM predictions of in-place runoff were used to estimate monthly streamflow at ungaged sites in the tributaries of the upper-middle and lower-middle Verde River between 1940 and 2006. The same BCM output (that included both in-place recharge and in-place runoff) used by Pool and others (2011) for initial estimates of recharge was used for this study.

NARGFM routes groundwater flow into streams, and model predictions can be directly used to estimate base flow (table 3). The Verde River, Oak Creek, Wet Beaver Creek, and West Clear Creek are represented in NARGFM with the MODFLOW Streamflow package (Prudic and others, 2004). The Streamflow package tracks inflow and outflow between a stream path and groundwater flow, and modeled predictions of base flow can be directly extracted from the Streamflow package output. Other tributaries, including Hell Canyon and Sycamore Creek, are not included in the current version of NARGFM. Adding other tributaries to NARGFM was beyond the scope of this project, but would be a straightforward process for studies focused on those areas.

Initial estimates of monthly streamflow were obtained by summing up BCM predictions of in-place runoff at each cell along stream paths defined by topographic gradients

Table 2. Numbers of model stream cells predicted by the Northern Arizona Groundwater Flow Model to be gaining, losing, or dry in reaches of the upper-middle and lower-middle Verde River, Arizona, in 1910 and 2006.

Type of streamflow	1910	2006	Difference	Percent difference
Gaining	721	664	–57	–8
Losing	549	552	3	0.5
Dry	348	402	54	14

Table 3. Base flows in cubic feet per second determined by the Northern Arizona Groundwater Flow Model and used in the monthly streamflow estimates for the middle Verde River watershed, Arizona.

Location	Base flow, in cubic feet per second
Oak Creek near Sedona gage	20
Oak Creek confluence	10
Wet Beaver Creek gage	1.5
Wet Beaver Creek confluence	1
West Clear Creek gage	17
West Clear Creek confluence	7.3

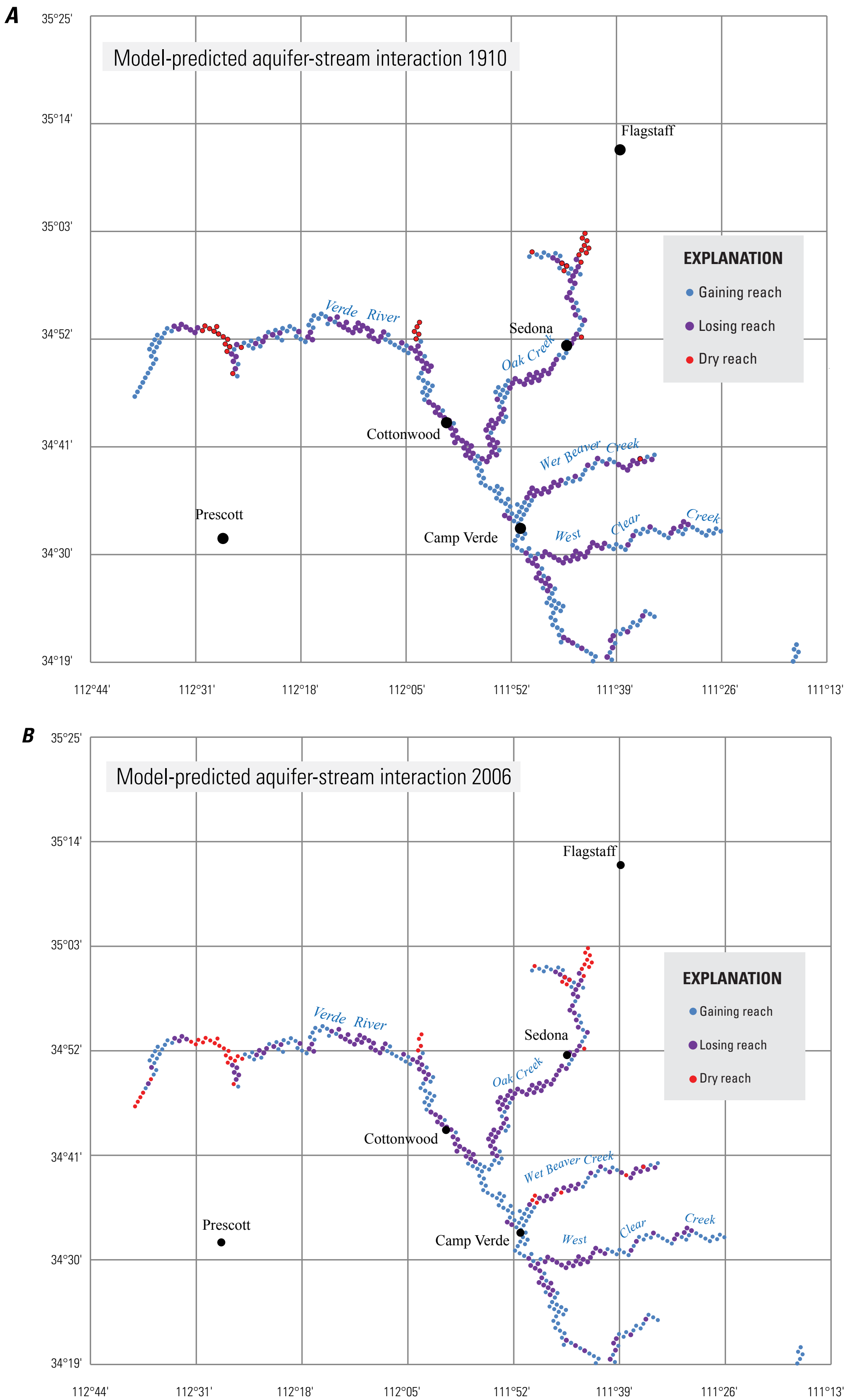


Figure 7. Latitude/longitude-gridded images of Northern Arizona Groundwater Flow Model predictions of groundwater-streamflow interactions on river reaches in the middle Verde River watershed, Arizona, showing reaches in (*A*) 1910 and (*B*) 2006 that were gaining discharge or losing discharge to groundwater and reaches that were dry.

using ArcGIS. Text files containing monthly BCM potential runoff data from 1940 to 2006 for the southwestern United States were converted to grids using a geographic information system (GIS) conversion tool. To decrease processing time, each potential-runoff grid was clipped to the study area boundary (fig. 8). A GIS flow-accumulation tool was then used to compute a weighted flow-accumulation grid for each month using a National Hydrography Dataset Plus flow direction grid (U.S. Environmental Protection Agency and the U.S. Geological Survey, 2006). A standard flow-accumulation grid is calculated using a flow direction grid to count the number of cells upstream of a given cell, giving each upstream cell equal weight. Using a potential-runoff grid to apply a weight to each cell in the flow-direction grid produces a weighted flow-accumulation grid that models a continuous surface model of accumulated potential runoff but does not account for interception, infiltration, or evaporation. A GIS tool to extract cell values at specified locations was then used to obtain accumulated potential-runoff data from the monthly weighted flow-accumulation grids for points of interest.

Various processes, such as the temporary storage of precipitation as snow or infiltration of runoff down-gradient, can affect the timing and magnitude of streamflow derived from the predicted in-place runoff. Because the BCM predicts in-place runoff and not streamflow, adjustment of the summed BCM predictions was required. The streamgages at Oak Creek near Sedona, and Wet Beaver Creek, Dry Beaver Creek, and West Clear Creek were used for the calibration. Gages on Granite Creek and Oak Creek near Cornville were not used because they are affected by reservoirs and diversions. The monthly gage data over the period of record minus the local base flow from NARGFM was plotted as a function of the BCM predictions at that gage location. Lines were fit to the plotted data for each month over the period of record and were used as corrections for the BCM predictions (table 4). A linear trend line constant of 0 was initially tried in all four locations and retained in three. A nonzero constant can cause a step change in calculated flow if the BCM predicts runoff for a particular month. The step change can be especially significant if the base flow is low or 0. Reasonable fits could not be achieved at Oak Creek near Sedona without a constant added, so the constant was included in the line equation for that site. In one case with considerable scatter in the relation between streamgage data and BCM predictions at Wet Beaver Creek, a linear fit produced poor results, and a power fit was substituted. During some months at some streamgages, there was insufficient flow in excess of base flow to fit a line, and the monthly predicted total streamflow is estimated to be base flow.

Adjusted BCM values were computed and the NARGFM-derived base flow added to form the predicted monthly streamflow from the combined NARGFM and BCM predictions (fig. 9). The Nash-Sutcliffe Efficiency Coefficient (Nash and Sutcliffe, 1970) was used to test the predictive power of the combined NARGFM and BCM results (table 5). This coefficient can vary between negative infinity and 1. Negative values indicate that the data mean is a better predictor than the model results. An efficiency of 1 indicates a perfect fit of the data to the model, whereas an efficiency of 0 indicates that the model results are as accurate as the data mean. An efficiency between 0 and 1 then indicates that the modeled values are positively correlated with measured values, and the closer the efficiency is to 1 the stronger the correlation. Nash-Sutcliffe Efficiency Coefficients for four tributaries varied from 0.49 to 0.72 (Dry Beaver Creek monthly streamflow was predicted from BCM only).

Table 4. Calibration coefficients used to relate combined Northern Arizona Groundwater Flow Model and Basin Characteristics Model predictions of local runoff to streamgage data for the middle Verde River watershed, Arizona.

[Fits are linear except where noted. The line constant is represented by *C*, the slope (or exponent in one case) is represented by *S*, and *R*-squared, the standard indicator of goodness-of-fit, R^2. Blank spaces indicate insufficient runoff was measured or predicted to generate a relation. *, Values represent the constant and exponent in a power function]

Month	Oak Creek near Sedona			Dry Beaver Creek			Wet Beaver Creek			West Clear Creek		
	C	*S*	R^2	*C*	*S*	R^2	*C*	*S*	R^2	*C*	*S*	R^2
1	-5.2	1.31	0.52	0	2.06	0.57	0	15.45	0.59	0	3.14	0.57
2	-5.8	1.45	0.8	0	1.77	0.66	0	1.38	0.15	0	1.39	0.75
3	57.48	0.58	0.59	0	1.03	0.49	82.11*	0.14*	0.23	0	0.75	0.47
4	75.76	0.07	0.1	0	0.4	0.16	0	0.32	0.15	0	0.31	0.32
5	4.8	0.03	0.59	0	0.11	0.44	0	0.08	0.51	0	0.1	0.51
6												
7												
8				0	3.63	0.94				0	1.13	0.75
9	11.07	0.59	0.86	0	4.43	0.85				0	3.69	0.22
10	8.03	0.07	0.34	0	1.04	0.93				0	1.88	0.77
11	-8.09	0.89	0.75	0	1.27	0.51				0	0.52	0.41
12	-59.38	2.38	0.7	0	2.73	0.29				0	2.41	0.43

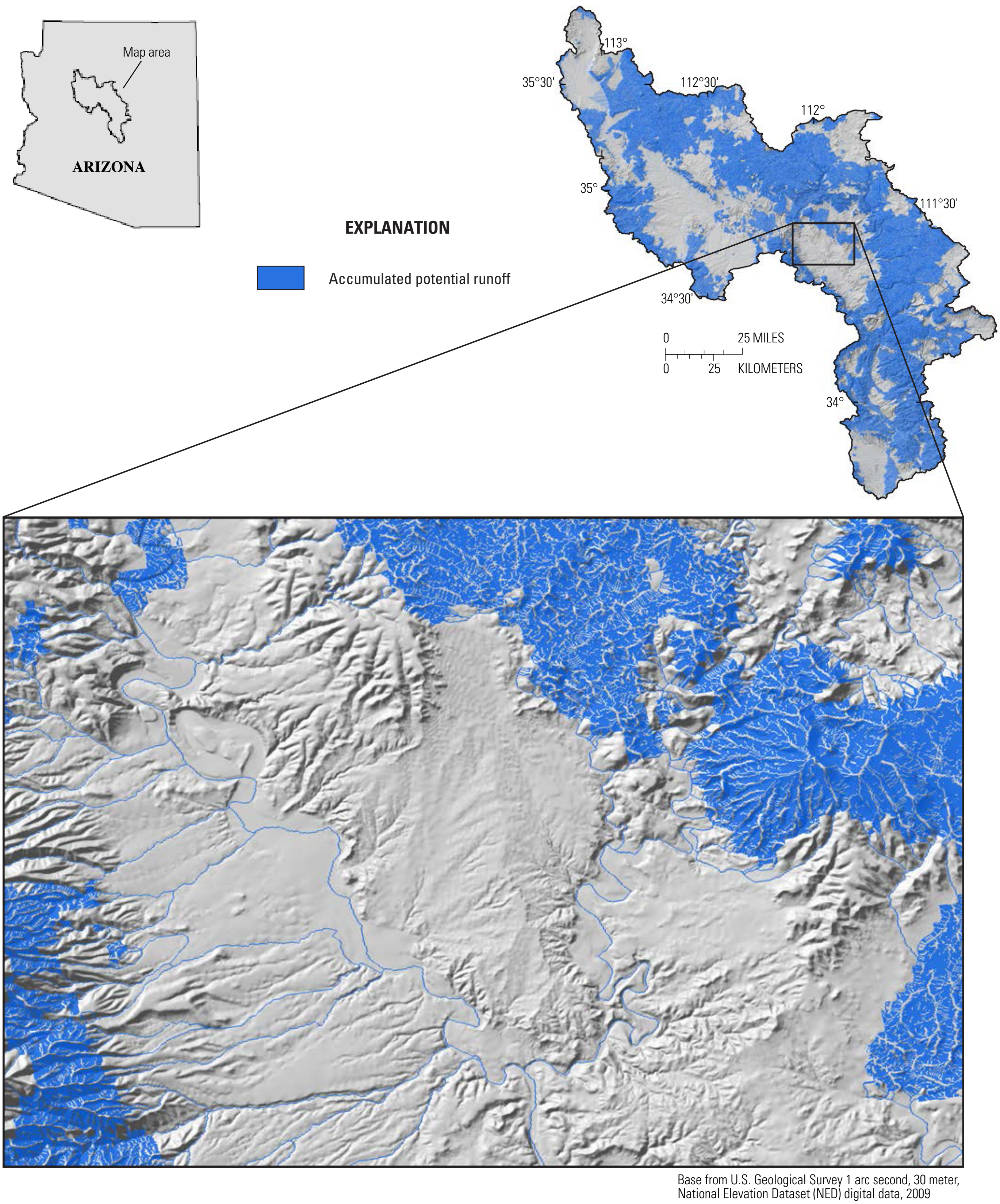


Figure 8. Map image showing weighted flow-accumulation grid generated from Basin Characteristics Model predictions of local runoff for February 1993 for the middle Verde River watershed, Arizona.

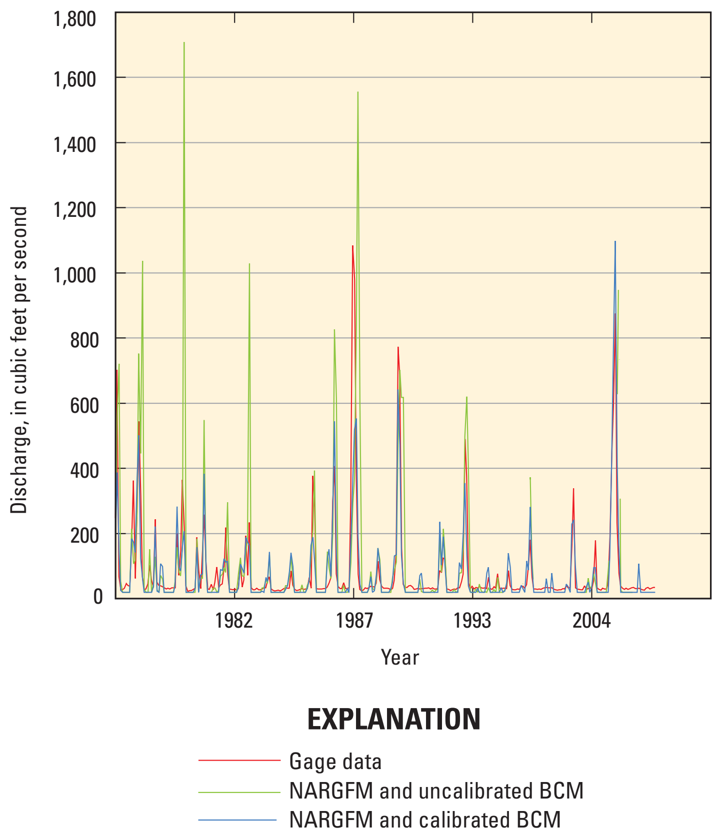


Figure 9. Example hydrograph showing monthly streamgage data at the Oak Creek near Sedona gage compared to combined Northern Arizona Groundwater Flow Model (NARGFM) base flow and uncalibrated Basin Characteristics Model (BCM) local runoff predictions and combined NARGFM base flow and calibrated BCM predictions.

Table 5. Nash-Sutcliffe efficiency coefficients for the fit of the combined Northern Arizona Groundwater Flow Model and calibrated Basin Characteristics Model predictions with respect to streamgage data for the middle Verde River watershed, Arizona.

Location	Coefficient
Oak Creek near Sedona	0.72
Dry Beaver Creek	0.66
Wet Beaver Creek	0.49
West Clear Creek	0.64

The combined NARGFM and calibrated BCM values were used to estimate total monthly discharges at ungaged sites in subbasins of the Verde River where the tributaries are included in NARGFM (Wet Beaver Creek and West Clear Creek), and the calibrated BCM values were used to estimate runoff values in tributaries that are not represented in NARGFM (Hell Canyon and Sycamore Creek) (fig. 1). The combined NARGFM base flows and calibrated BCM runoff predictions of monthly streamflow and streamgage data were used to compute exceedance discharges at the 80th, 50th, and 20th percentiles (figs. 10–13). The computed values generally show better agreement with streamgage data at the two streams with higher base flow and discharges (Oak Creek near Sedona and West Clear Creek). With the accumulations of BCM predicted local runoff and with monthly calibration functions, smaller-scale basin monthly runoff could then be estimated at locations of interest (fig.14).

Hydrologic simulation resolution is always a factor in assessing streamflow alteration and determining the environmental flows appropriate to the scale of the measured ecology. Regional-scale models at the monthly time step can assist with regard to climate-scale impacts on streamflow and species distributions, whereas high-resolution models can address effects to reach-scale assemblages. Both can be leveraged to provide a comprehensive picture of changing ecosystems, while balancing the conservation of water for long-term human needs (Caldwell and others, 2015). Regardless of the approach, the hydrologic simulation is only as accurate as the hydrogeological understanding and the surface-water, groundwater and climate data that go into the model. The long time step of the BCM and NARFGM provide stream averages that cannot capture the variability in magnitude and duration of runoff events, particularly in watersheds with highly variable discharge such as that of the Verde River. Many processes relevant to in-steam ecology and thus environmental flows more generally, such as sediment transport, involve discharge thresholds and nonlinear responses to streamflow. Many other regions in United States can use monthly streamflow estimates in environmental flows studies (Herkinson and others, 2006; Archfield and others, 2014; Kennen and others, 2014), but the constraints of this ecological dataset require streamflow data recorded at daily or shorter time steps in order to relate streamflow to in-stream habitat.

Riparian Vegetation

Riparian forests (forests along river corridors) occupy less than 2 percent of the land area in the southwestern United States and less than 1 percent of the desert landscape but are generally considered the most productive and valuable land in the region (Folliott and others, 2004; Kirkpatrick, 2008). Riparian areas are critically important for wildlife (DeBano and Schmidt, 2004). For example, riparian woodlands support more than 50 percent of the breeding birds in the desert southwest (Kirkpatrick and others, 2008). The Verde River

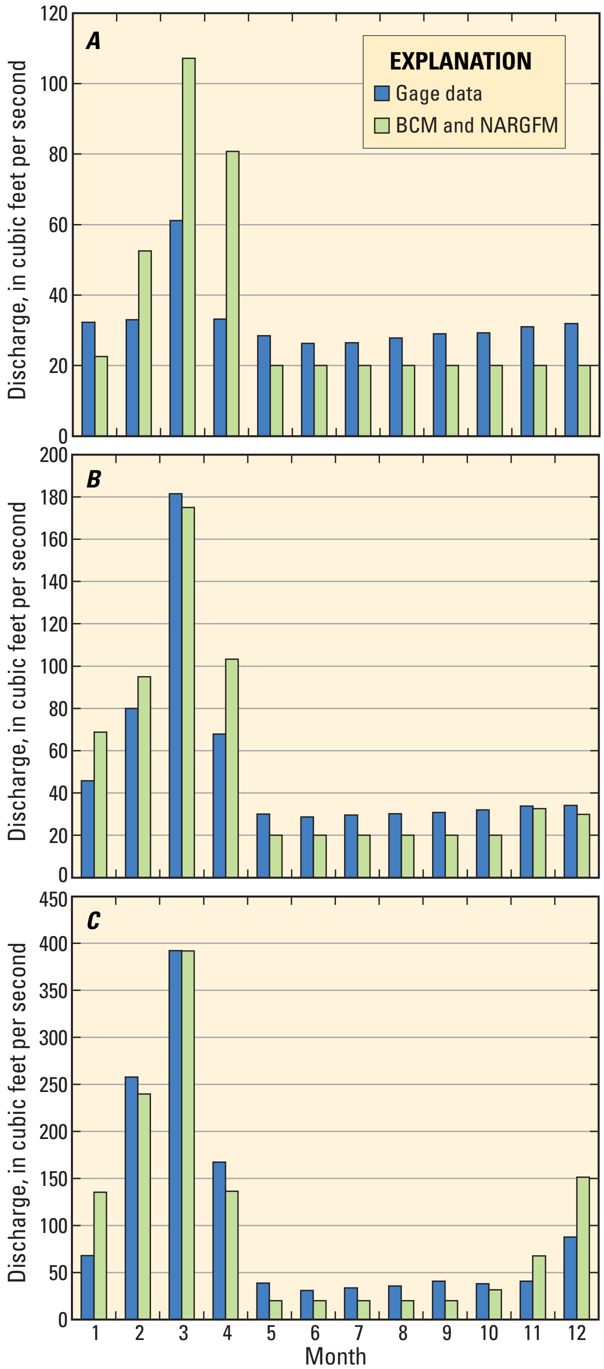


Figure 10. Graph showing monthly streamflow statistics at the Oak Creek gage in the middle Verde River watershed, Arizona, determined from gage data and combined Basin Characteristics Model (BCM) and Northern Arizona Groundwater Flow Model (NARGFM) predictions. The plots show the (*A*) 80-, (*B*) 50-, and (*C*) 20-percent monthly exceedance discharges.

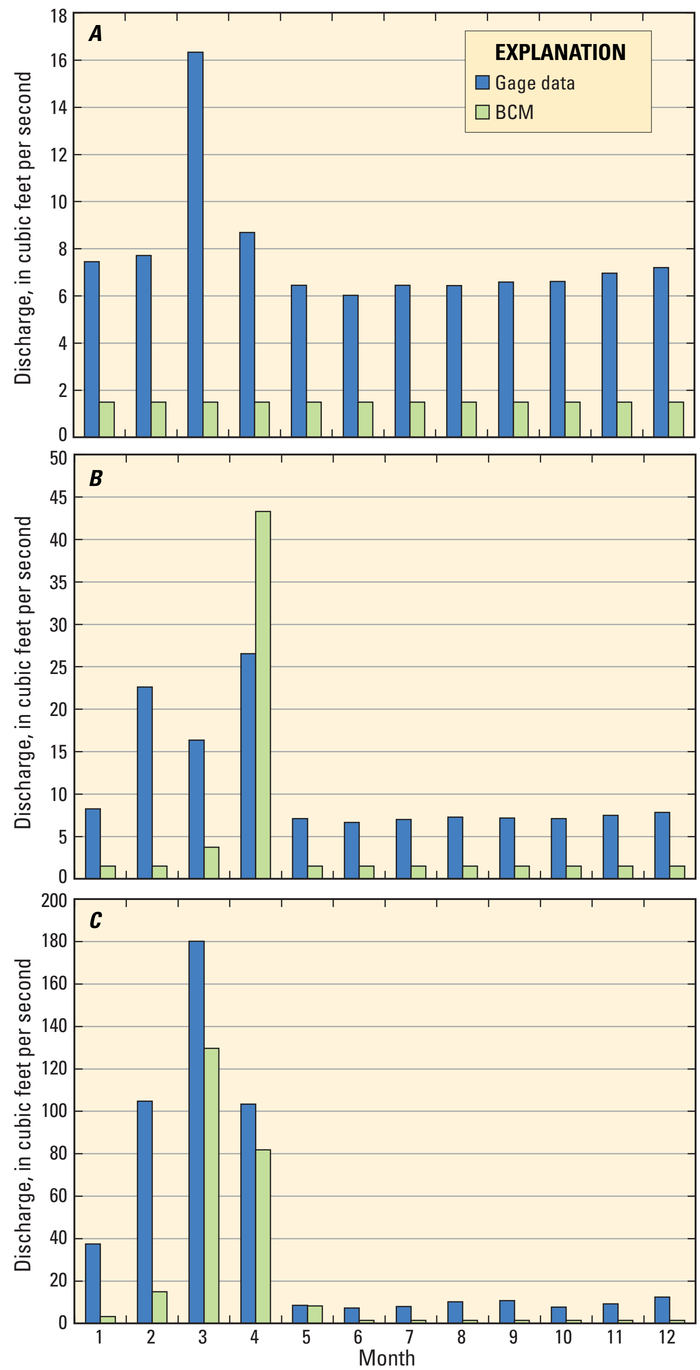


Figure 11. Graph showing monthly streamflow statistics at the Wet Beaver Creek gage in the middle Verde River watershed, Arizona, determined from gage data and combined Basin Characteristics Model (BCM) and Northern Arizona Groundwater Flow Model (NARGFM) predictions. The plots show the (*A*) 80-, (*B*) 50-, and (*C*) 20-percent monthly exceedance discharges.

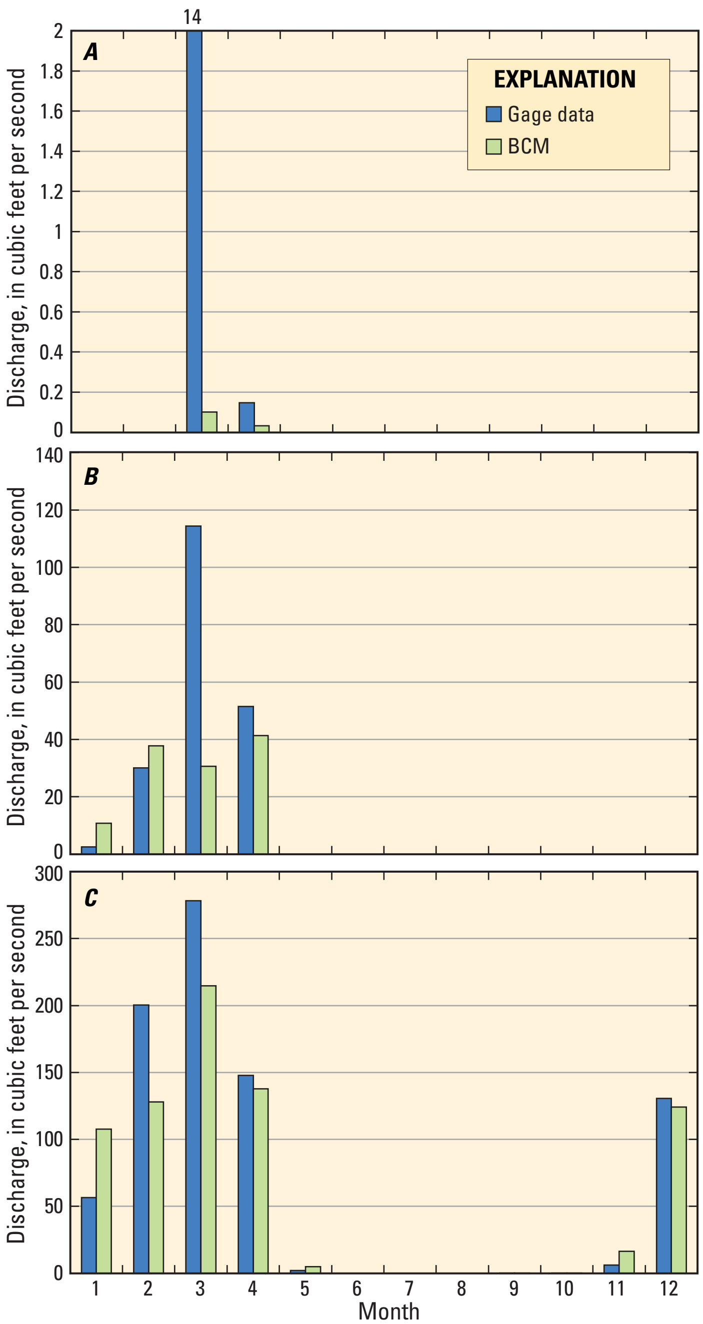


Figure 12. Graph showing monthly streamflow statistics at the Dry Beaver Creek gage in the middle Verde River watershed, Arizona, determined from gage data and Basin Characteristics Model (BCM) predictions. The plots show the (*A*) 80-, (*B*) 50- , and (*C*) 20-percent monthly exceedance discharges.

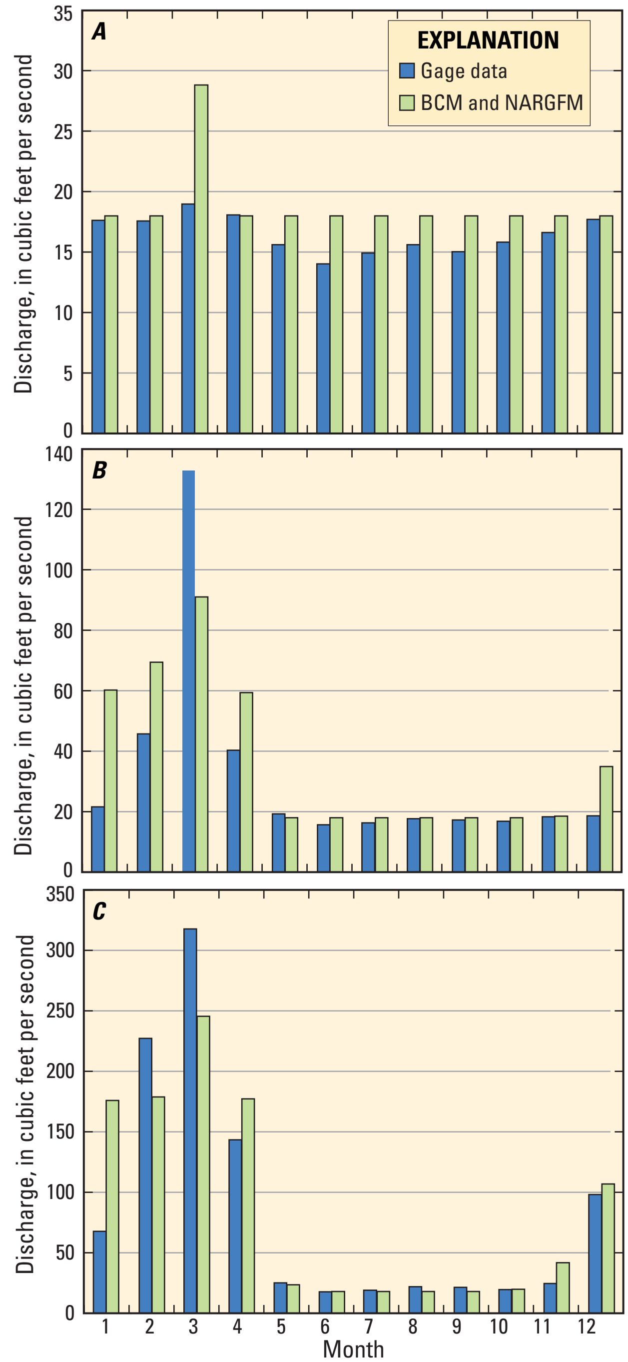


Figure 13. Graph showing monthly streamflow statistics at the West Clear Creek gage in the middle Verde River watershed, Arizona, determined from gage data and combined Basin Characteristics Model (BCM) and Northern Arizona Groundwater Flow Model (NARGFM) predictions. The plots show the (*A*) 80-, (*B*) 50-, and (*C*) 20-percent monthly exceedance discharges.

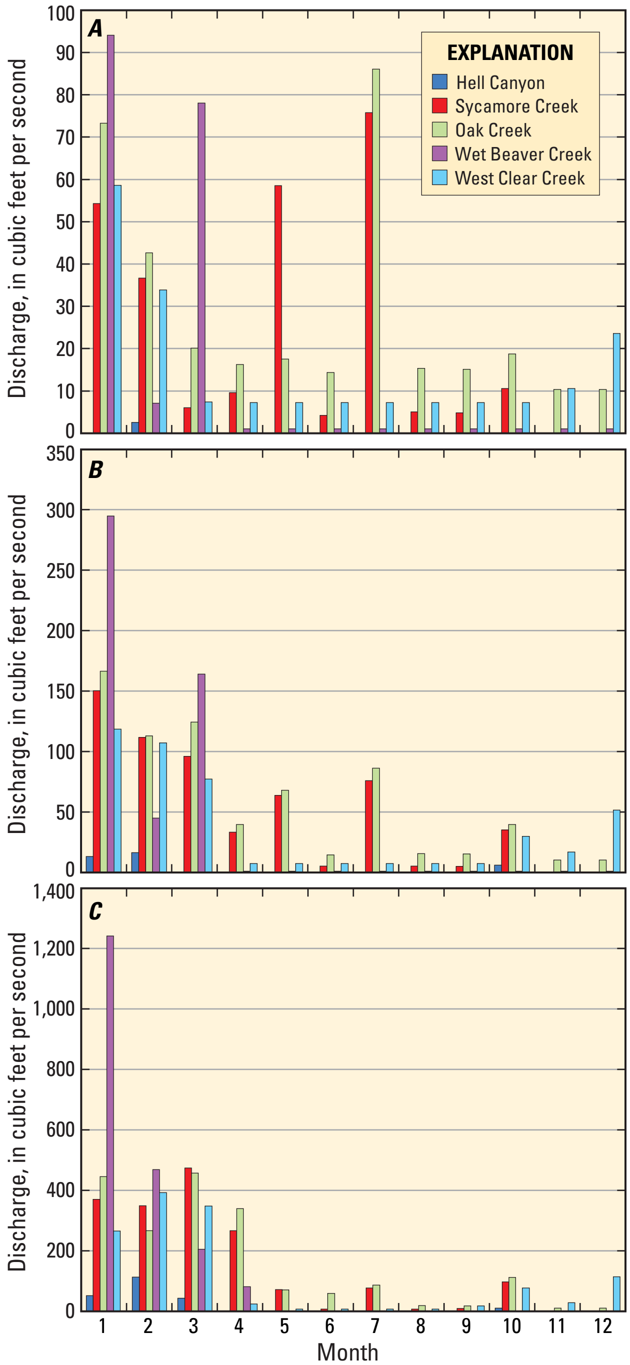


Figure 14. Graph showing monthly streamflow statistics at tributary confluences in the middle Verde River watershed, Arizona, determined by Northern Arizona Groundwater Flow Model (NARGFM) predictions of base flow, where available, and calibrated Basin Characteristics Model (BCM) predictions of runoff. The plots show the (*A*) 80-, (*B*) 50-, and (*C*) 20-percent monthly exceedance discharges.

watershed contains some of the most extensive acreage of Fremont cottonwood (*Populus fremontii*)-Gooding willow (*Salix gooddingii*) forest—found only in the southwestern United States and Mexico—and mixed broadleaf riparian forest in Arizona. The current, relatively unregulated flow of the Verde River means the riparian vegetation is of mixed ages. Mixed-age stands are instrumental to increasing forest resiliency and act as key areas where studies can be conducted on the conditions needed for tree establishment. The Verde River watershed is home to special-status species such as the yellow-billed cuckoo and the southwestern willow flycatcher (Stevens and others, 2008). The plant communities of the Verde River watershed that support these rare birds are subject to the same stresses that are changing plant communities throughout the Southwest, such as increasing pressures from agriculture and population density, as well as small-scale river regulation through the use of irrigation ditches. Work on other streams has shown that changing the natural flow regime of a river affects both aquatic and riparian species (Poff and others, 1997; Stromberg and others, 2007).

Riparian forest response to streamflow regime and depth-to-groundwater fluctuations have been extensively studied in southwestern rivers. Reduced base flow is known to alter aquatic habitat, with some reaches changing from perennial to intermittent (Haney and others, 2008). In addition, the average annual depth from the land surface to saturated soils typically increases as base flow declines. Under these conditions it is hypothesized that there will be declines in willow and cottonwood abundance, decreases in structural diversity, and increases in non-native species such as tamarisk (Stromberg, 1993, 2008; Shafroth and others, 1998; Stromberg and others, 2007). Such vegetation changes would likely cause shifts in the bird community, with reductions or loss of some species.

The main woody riparian species found within the study area (described below) included Fremont cottonwood, Goodding's willow, Arizona cypress (*Cupressus arizonica*), Colorado pinyon pine (*Pinus edulis*), Utah juniper (*Juniperus osteosperma*), and mesquite forests of velvet mesquite (*Prosopis velutina*), screwbean mesquite (*Prosopis pubescens*), and honey mesquite (*Prosopis glandulosa*). Cattail (*Typha* spp.) were also prevalent in the study area. Non-native species including Russian olive (*Elaeaganus angustifolia*), giant cane (*Arundo donax*), tamarisk or salt cedar (*Tamarix ramosissima*), and tree of heaven (*Alianthus altissima*) also occur in the study area.

Life Histories and Ecological Needs of Cottonwood and Willow Species

Individual trees in cottonwood-willow gallery forests that grow along low-elevation rivers in the southwestern United States typically have a life span of 100 to 150 years (Stromberg, 1993). Cottonwood and willow trees are obligate phreatophytes–they have deep roots and will only grow in areas where they have constant access to groundwater (DiTomaso, 1998). Both cottonwoods and willows are also pioneer species, meaning they can colonize disturbed areas; flooding is the main disturbance in the areas where they grow. Both species are prolific seed producers, reproducing mostly by seeds rather than asexually, although both species can sprout shoots from buds on branches laid out by floods. Germination is timed to coincide with large flood events during the spring and early summer, which helps seedlings to establish above the zone of frequent subsequent flooding (Stromberg, 1993). Recruitment (seedling survival to maturity) typically occurs once every 10 to 15 years with some geographic variability. Both Fremont cottonwood and Goodding's willow seeds are viable for 1 to 5 weeks after dispersal and germinate in 24 to 48 hours once proper conditions have been met (moist, unvegetated mineral soil or alluvium) (Fenner and others, 1984, 1985; Stromberg, 1993). Drought and summer and fall floods are the main causes of mortality for both cottonwood and willow seedlings (Stromberg, 1993). Several studies have suggested that a water-table drop of about 1.5 meters (m) can result in adult cottonwood mortality (Stromberg and others, 1996; Scott and others, 1999; Shafroth and others, 2000).

Non-native Riparian Species

Non-native riparian species compete with native species for sites with access to water and can cause broader changes in the riparian community. The most abundant non-native tree species on the Verde River was Russian olive. Russian olive was the most abundant non-native tree in this area probably due to the higher elevation in the upper-middle Verde River, resulting in cold winter temperatures that meet Russian olive's chilling requirement for seed germination and bud break (Guilbault, 2011; Shafroth and others, 2010a). The phenological (seasonal lifecycle) and reproductive needs of Russian olive are not well understood; however, it is known to spread rapidly once established, competing for space with native vegetation. As a result, Russian olive is a threat to the rare gallery forests of the Verde River and is thus a significant management concern.

Other major vegetation-management concerns include giant cane, tamarisk or salt cedar, and tree of heaven. Relative percentages of all the different non-native species are currently unknown, although the Friends of the Verde River Greenway have done some mapping of their location and extent along specific reaches of the river (Fred Phillips Consulting, 2011). Tamarisk on the Verde River was found to make up about 8 percent of the total plant community, which is a relatively small percentage compared to the extent of tamarisk's range in other western riparian corridors (Johnson and others, 2009).

Tamarisk is a halophyte, a drought- and salt-tolerant plant that can complete its entire life cycle under saline conditions. Some work suggests that tamarisk can negatively affect aquatic macroinvertebrates (Bailey and others, 2001; Webb and others, 2007). Both cottonwood and willow trees are glycophytes—they prefer soils with minimal salt content. Past studies have shown these species experience upwards of 50-percent reduction in growth when soil salinity reaches 5 grams per liter (g/L) (Glenn and others, 1998; Shafroth and others, 2010a). Johnson and others (2009) reported that in the upper-middle Verde River watershed, areas where tamarisk grew had higher abundance of all plant species than areas where tamarisk did not grow. This suggests that tamarisk may be subjected to the same limiting factors to growth as other plants. In the study area, tamarisk has been noted growing in the cottonwood understories (beneath the forest canopy), which is not a typical pattern for the species (Shafroth and others, 2010a).

Cottonwood trees have been shown to have complex symbiotic relations with mycorrhizal fungi that grow on their roots and assist in nitrogen fixation in exchange for other nutrients (Beauchamp and others, 2005). Beauchamp and others (2005) found that tamarisk grown in pots with cottonwood had a 15-percent reduction of pot biomass when inoculated with mycorrhizal fungi; this may tend to limit tamarisk growth near established cottonwoods.

Giant cane is part of the Arundineae family thought to be native to eastern Asia. It is one of the largest grass species and is abundant along the Verde River, especially along Oak Creek. It is a clonal plant that grows densely and can out compete other plants. The non-native species group of the World Conservation Union lists giant cane in the top 100 "worst invaders of the world" (Lowe and others, 2000). Giant cane is a hydrophyte; it grows best near water and is adapted to living in waterlogged soil (California Invasive Plant Council, 2011). Under laboratory settings, it was found that leaf defoliation or leaf damage did not significantly affect or alter the growth of giant cane (Spencer, 2012). Potential vegetative destruction that results from the tamarisk leaf beetle (*Diorhabda* spp.) and biological controls for Russian olive (the subject of U.S. Department of Agriculture research) may create more ecological and physical room for other non-native plants such as giant cane (rather than for just native species); it is thus possible that giant cane could become more widespread in the middle Verde River watershed in the future.

Assessment of Current Riparian Vegetation Status

Division of Reaches

The physical area of the vegetation study extended from river mile 191 at the headwaters of the Verde River (below Granite Creek confluence) to mile 109, just below Camp Verde. Analysis was also conducted on two tributaries below Clarkdale—Oak Creek, where the vegetation survey was conducted at a transect 17.5 miles upstream from the confluence with the Verde River, and West Clear Creek, where analysis was conducted at a transect 11 miles upstream from its confluence with the Verde River.

For the purposes of the vegetation analysis, the study area on the Verde River was divided into two sections, one above and the other below the town of Clarkdale, Arizona (upper-middle reach and lower-middle reach, respectively). This designation was made because of the differences in land use, elevation, and sinuosity between the upper and lower section of the study area. The upper-middle reach is from Sullivan Dam, near the Verde headwaters, to the town of Clarkdale, covering river miles 191 to 144. In the upper-middle reach, the river starts in a broad floodplain, then enters narrow, winding canyons and has nine tributaries entering the main stream. Most of the human population along the upper-middle reach is near Chino Valley, and there is little habitation along the rest of this section.

The lower-middle reach extends from the town of Clarkdale to the town of Cottonwood, covering river miles 143 to 109. Below Clarkdale, the river is wider than in the upper-middle reach. There are eleven tributaries along this reach and the towns of Clarkdale, Cottonwood, and Camp Verde are located here, making the lower-middle reach the more populated of the two study reaches. The general land uses in this reach are agriculture, recreation, and residential (Hazelton, 2011). Some of the agricultural land has been converted to private residences that retain their agricultural water right.

Remote Sensing of Evapotranspiration

Remote sensing of evapotranspiration (ET) is a useful tool for land managers interested in vegetation change through time. Products from the Moderate Resolution Imaging Spectroradiometer (MODIS) and Landsat sensors provide free satellite imagery and were used for this study. Collection of ground data for calculating ET or leaf area index on the Verde River were beyond the scope of this study. Measurements such as the enhanced vegetation index (EVI) are effective measurements of plant vigor in the study area. This study uses the MODIS instrument on the Aqua satellite to calculate EVI, which is then used to calculate ET (fig. 15*A* and *B*). EVI is a measure of greenness but has the advantage of not getting saturated at higher levels of greenness (Huete and others, 2002). At the time of download, MODIS EVI data (MOD13Q1 from Oak Ridge National Laboratory, http://daac.ornl.gov/) were available from day of year (DOY) 49 of 2000 to DOY 65 of 2012.

Imagery

Orthoimagery was obtained from Yavapai County in 2010 for the area of the Verde River that extends from just above Clarkdale to Sycamore Creek East and in 2007 for Oak Creek and Wet Beaver Creek. The resolution of the Yavapai County orthoimagery is 0.5 feet (ft). Dense areas of vegetation were assumed to be mixed stands of cottonwood, willow species, and possibly non-native plants such as tamarisk. For areas of the Verde River north of Clarkdale and West Clear Creek, where imagery was not available from Yavapai County, aerial imagery was obtained by importing Bing Maps into ArcMap 10. These aerials have slightly coarser resolution and a different projection than the Yavapai County aerials, and digitizing methods were adjusted accordingly. The time of year these aerial images were obtained is unknown. Based on the greenness of vegetation species such as mesquite, it is assumed that the images were taken during a time of year with rapid growth. Deep, incised channels with high, steep banks of some parts of the upper-middle Verde River caused much of the vegetation to be shaded, reducing accuracy when digitizing those areas.

Ground-truthing was conducted at Tuzigoot National Monument (along the Verde River near Clarkdale), Dead Horse Ranch State Park (along the Verde River near Cottonwood), and Campbell Ranch (near the Verde River headwaters). Ground-truthing was done in July, September, and December of 2011 by identifying plant species, marking their locations with a Global Positioning System (GPS) and then importing those points into ArcMap and comparing the ground-identified vegetation with the map imagery.

Vegetation Digitizing and Classification

Vegetation along the Verde River in the study area was identified from the imagery within a buffer extending out 328 ft (100 m) from both sides of the wetted channel. At locations where the channel split, one polygon encompassed both branches of the wetted channel. Seven vegetation classifications were used—mesquite, cottonwood, mixed canopy, mixed shrub, pinyon-juniper, agriculture, and emergent aquatic vegetation (table 6).

Table 6. Vegetation classes used to digitize satellite and aerial imagery along the Verde River, Arizona.

Vegetation class	Description
Mesquite (*Prosopis* spp.) dominant	Dead and brushy appearance; may include other shrubs and subtrees.
Cottonwood (*Populus* spp.) dominant	Tallest trees observed with large, readily identifiable shadows and distinctive shapes.
Mixed canopy	Mixed species growing in close proximity, including willow (*Salix* spp.). Includes trees that could not be accurately identified individually.
Mixed shrub	Appear similar to mesquite trees but smaller and of mixed species.
Pinyon-juniper (*Pinus* spp. and *Juniperus* spp.)	In the uplands areas, circular and dark green.
Agriculture	Includes both planted and fallow fields; both were assumed to be in use.
Emergent aquatic plants	Plants near the edge of the wetted channel.

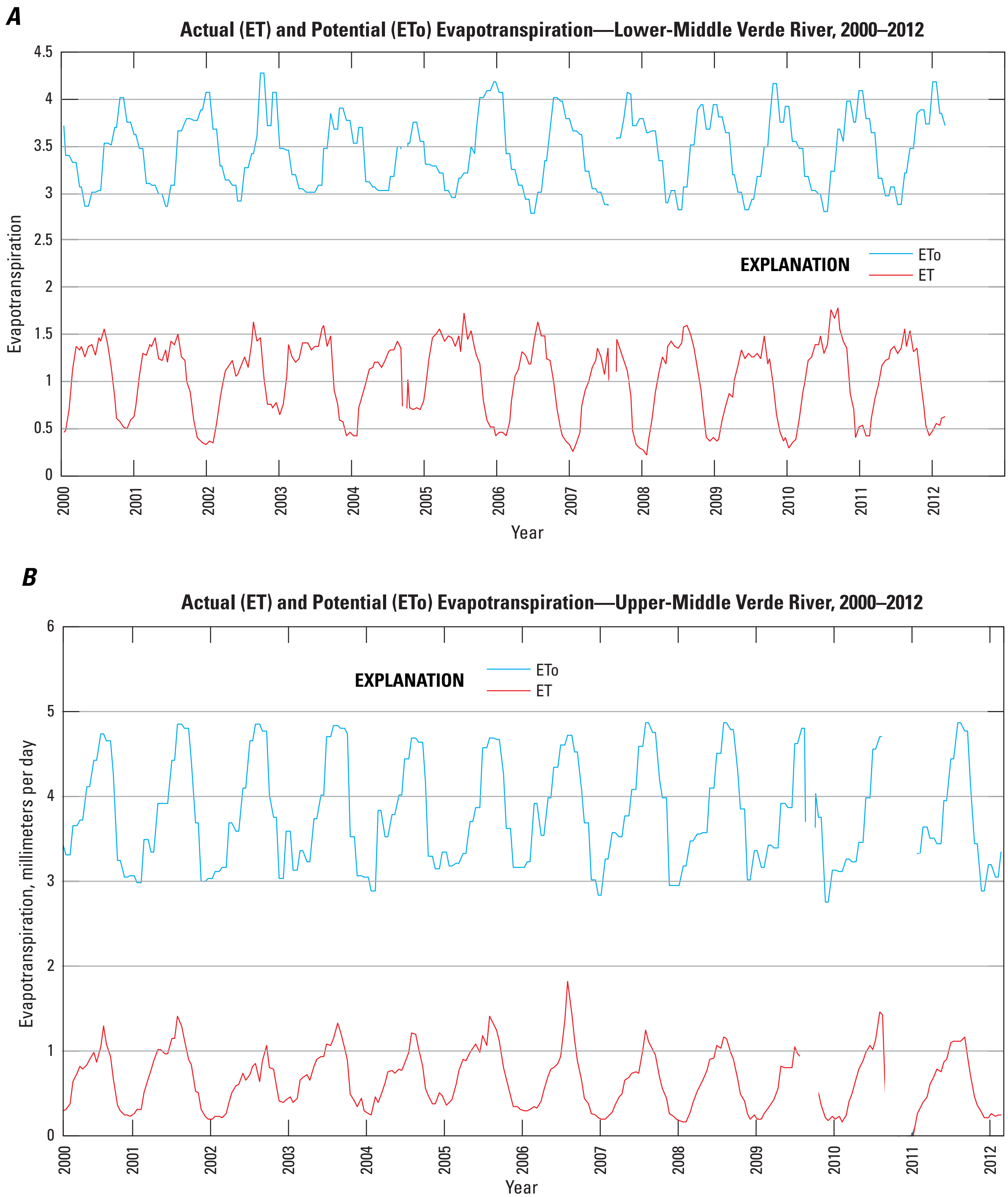


Figure 15. Graphs showing actual and potential evapotranspiration of the Verde River, Arizona, from 2000–2012 using the Moderate Resolution Imaging Spectroradiometer (MODIS) enhanced vegetation index (EVI) product. *A*, Lower-middle Verde River; *B*, upper-middle Verde River.

Mesquite and cottonwood have separate classes because there is enough confidence in their distinctive shapes and greenness at the time images were flown to specifically identify them. Because of the current, relatively unregulated flow of the Verde River, the riparian vegetation is of mixed ages. This is important in class designation because younger cottonwood trees did not have the distinctive canopies of the older cottonwood trees digitized in the Cottonwood class and are thus accounted for in the mixed-canopy class. Other species in this class are the many species of willow and the other trees, such as the non-native tamarisk, that we were not able to identify separately on the aerial imagery.

The mixed-shrub and the mesquite classes are the only vegetation classes that include bare soil between plants. This is because the increased irregularity in the size of the mixed-shrub and mesquite vegetation with similarly irregular patches of bare soil in between plants made it inefficient to exclude bare soil, but likewise inefficient to make bare soil its own class. A fifth vegetation class was added for the pinyon-juniper vegetation type that is prevalent in the upper reach. The pinyon-juniper stands have a distinctive circular shape, are green year round, and grow above the riparian zone at higher elevations including the upper part of the river. The sixth vegetation class is agriculture and the seventh class is for emergent aquatic vegetation. In some river miles, the percentages of vegetation add up to less than 100 percent. For these areas, the remaining percentage is either bare soil or developed land, neither of which were digitized.

Figures 16 and 17 show the distribution of vegetation classes along the upper-middle (fig. 16) and lower-middle (fig. 17) Verde River. As would be expected, due to the higher elevation and narrower channel, the upper-middle section has more upland plant species, such as pinyons and junipers, and fewer lowland species, such as cottonwoods, compared to their abundance in the lower-middle sections (figs. 16 and 17). Differences in vegetation within the upper-middle and lower-middle reaches are more likely related to differences in climate and hydrology than to human alterations. Both Oak Creek (fig. 18) and West Clear Creek (fig. 19) are dominated by mixed shrub and mixed canopy, but Oak Creek has a greater percentage of cottonwoods and mesquite than West Clear Creek. This may be in part because of the habitat provided by the steeper and narrower active channel of West Clear Creek, as well as its predominantly east-west orientation.

Because of the relatively unregulated flow and relatively unaltered disturbance regime, particularly of the upper-middle Verde River watershed, the riparian vegetation is of a mixed age, mixed patch structure, which is a sign of a healthy ecosystem (Shafroth and others, 2010b). Alteration of hydraulic regime can negatively affect the current cottonwood-willow gallery forest, as well as any future forest establishment by preventing seedling establishment, lowering the groundwater levels needed to maintain the current vegetation, and by making non-native expansion possible into areas where native plants can no longer survive due to decreased groundwater levels. Continued reduction of base flows in the Verde River also has the potential to facilitate the establishment of additional non-native plant species and to expand the current populations of tamarisk, Russian olive, tree of heaven, and giant cane.

Geomorphology

The physical setting of in-stream habitat is defined by geomorphic processes. Channel planform, depths, and velocity distributions are largely determined by valley relief, sediment supply and caliber, and discharge. Pearthree (1993, 1996, and 2008) described the Verde River geomorphology with an emphasis on the ages of channel deposits and includes a comparison of channel positions over time taken from aerial photos. Using the categories of Rosgen (1996), Neary and others (1996) classified reaches of the upper Verde River in relation to riparian habitat. Moody and others (2003) estimated bank full discharge and bank stability through cross-sectional surveys. Pearthree (2008) noted the predominance of coarse bed material and pool-riffle-rapid sequences, that low-flow and flood channels are typically well defined, and that riffles form at low flow around alternating gravel bars but are washed out at higher discharges.

Sullivan Dam, located near Paulden, is upstream of the confluence of the Verde River and Granite Creek. The effect of Sullivan Dam on sediment transport through the Verde River is unknown. The reservoir behind the dam filled with sediment, but the reduction in slope upstream of the dam would tend to promote deposition of sediment and thereby limit sediment supply below the dam. The extent to which this is occurring now or whether the effect has been significant in the past is unknown. Steep-gradient tributaries draining the highlands continue to supply sediment as is evident in the coarse deposits at the confluence of the Verde River with Granite Creek and in the sand-filled pools observed below the Granite Creek confluence. Bed material may also be supplied by bank and overbank material. Pearthree (2008) distinguishes between channel alluvium based on age and notes that the younger deposits are more susceptible to scour.

Geomorphic Classification of River Features

Water depth and velocity, which are controlled by channel geomorphology, can be determining factors in the suitability of in-channel habitat for certain fish and macroinvertebrates. In the middle Verde River, geomorphic units were determined from observations made directly from orthoimagery. Because the Yavapai County orthoimages were taken in October, before snowmelt, the stream is assumed to have been close to base flow. Three types of aquatic habitat features were classified—riffles, runs, and pools (Arend, 1999). Methods used to characterize the water surface features are modified from Konrad and others (2008a). Classes of riffle, run, and pool (table 7) were determined in areas of high resolution on

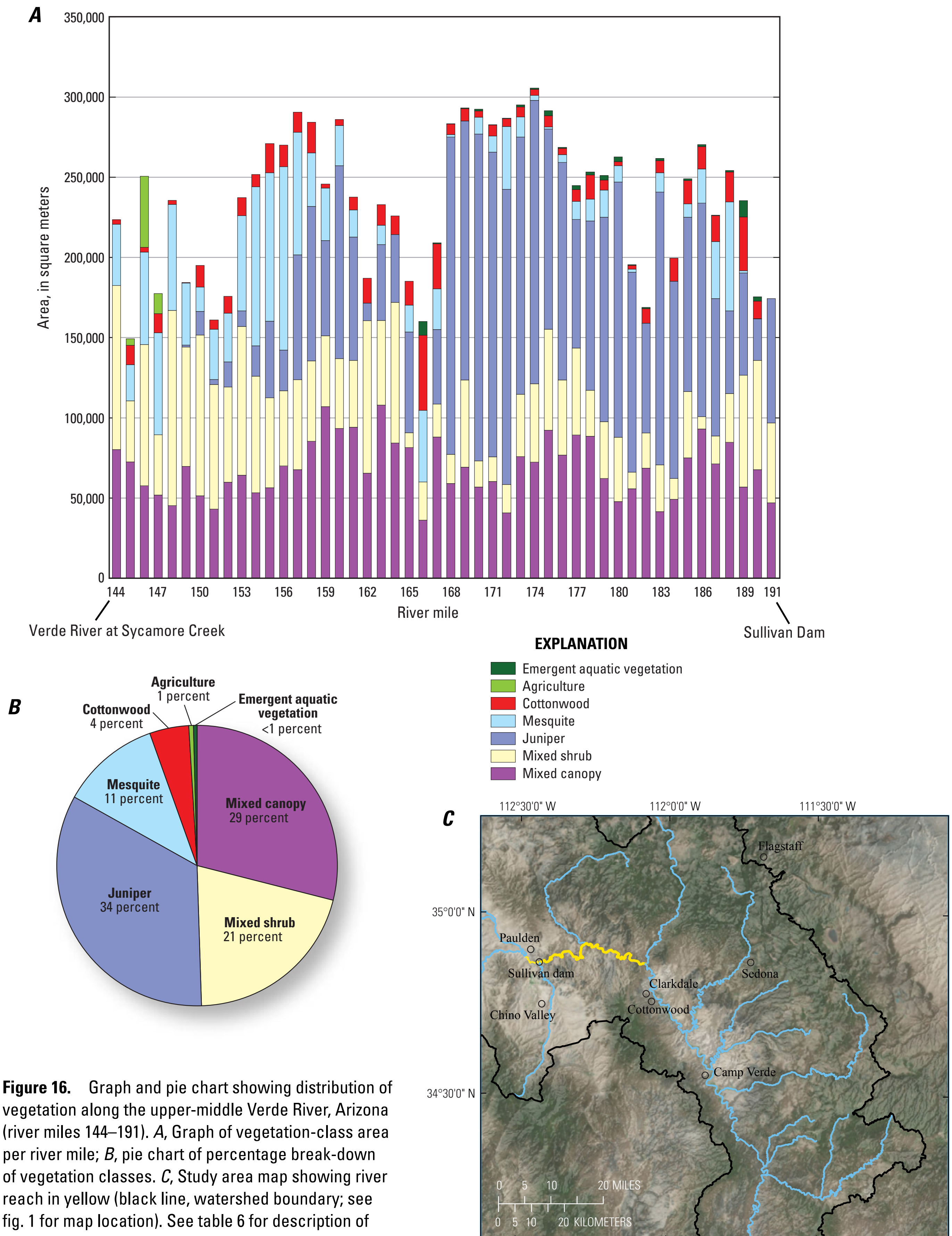


Figure 16. Graph and pie chart showing distribution of vegetation along the upper-middle Verde River, Arizona (river miles 144–191). *A*, Graph of vegetation-class area per river mile; *B*, pie chart of percentage break-down of vegetation classes. *C*, Study area map showing river reach in yellow (black line, watershed boundary; see fig. 1 for map location). See table 6 for description of vegetation classes.

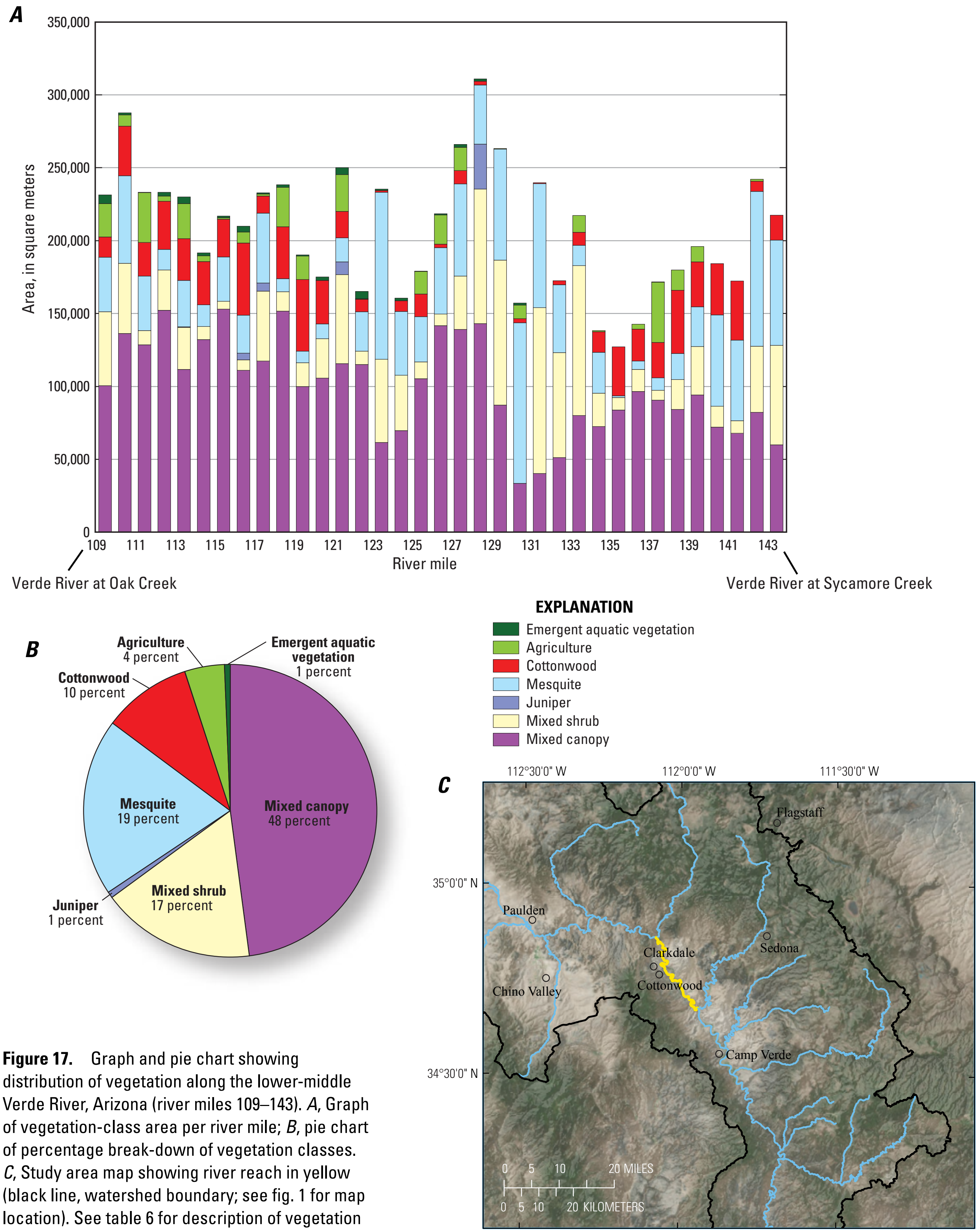


Figure 17. Graph and pie chart showing distribution of vegetation along the lower-middle Verde River, Arizona (river miles 109–143). *A*, Graph of vegetation-class area per river mile; *B*, pie chart of percentage break-down of vegetation classes. *C*, Study area map showing river reach in yellow (black line, watershed boundary; see fig. 1 for map location). See table 6 for description of vegetation classes.

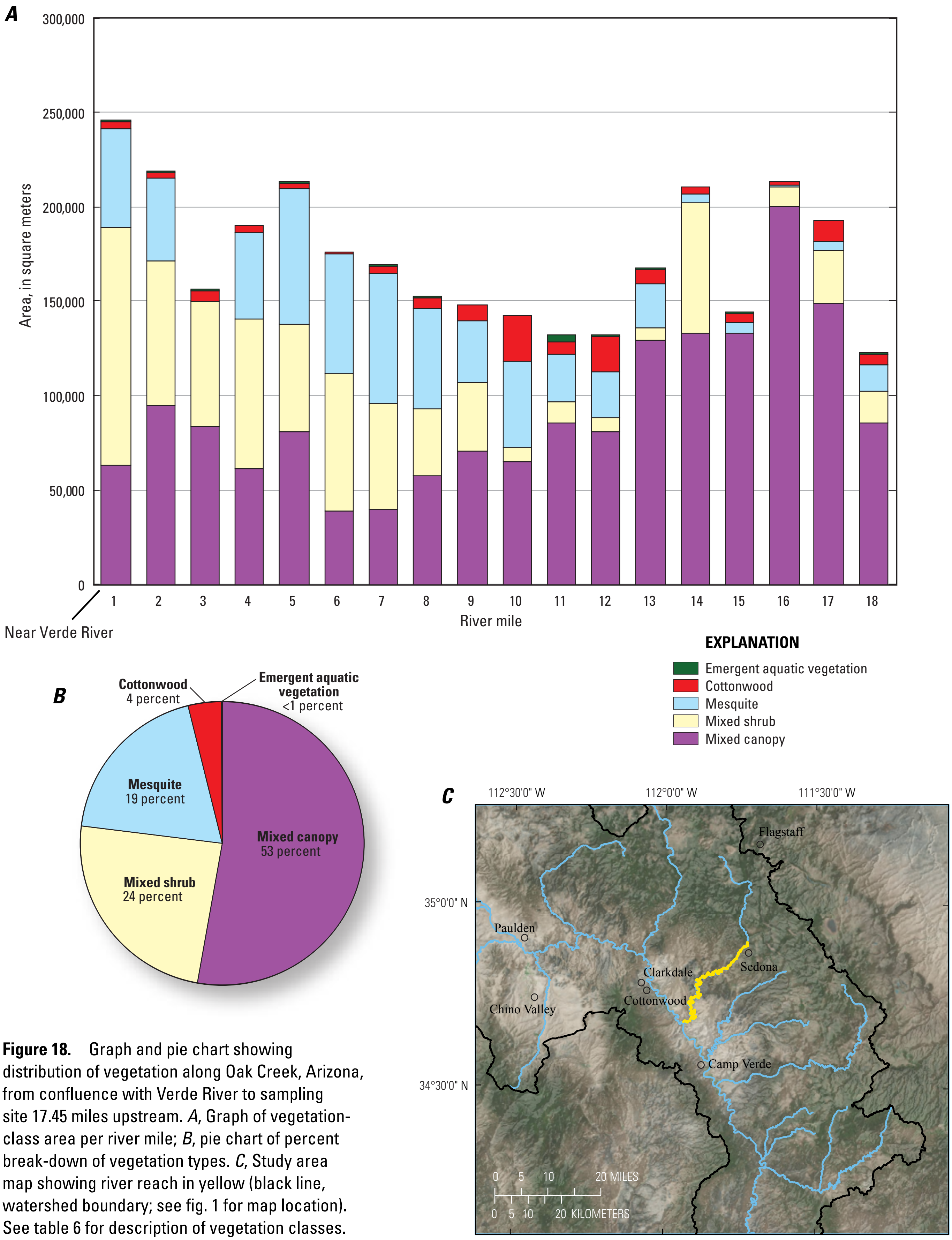
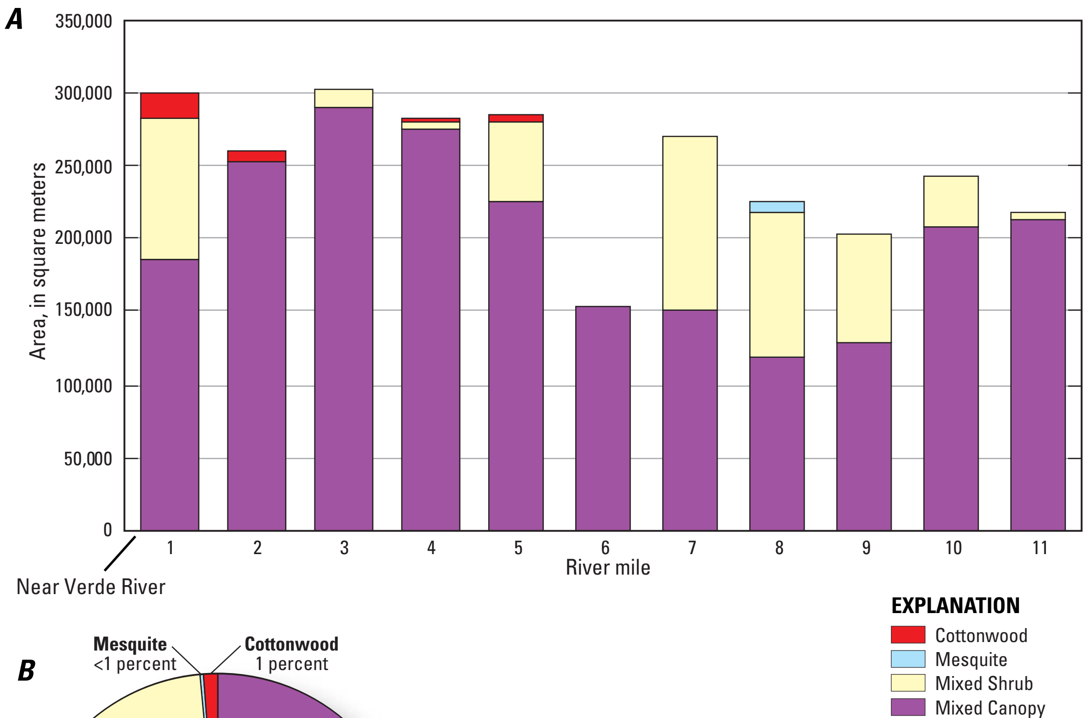
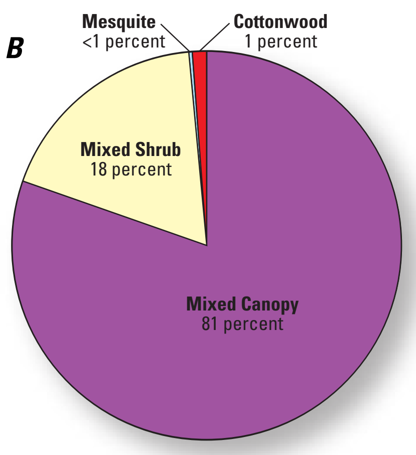


Figure 18. Graph and pie chart showing distribution of vegetation along Oak Creek, Arizona, from confluence with Verde River to sampling site 17.45 miles upstream. *A*, Graph of vegetation-class area per river mile; *B*, pie chart of percent break-down of vegetation types. *C*, Study area map showing river reach in yellow (black line, watershed boundary; see fig. 1 for map location). See table 6 for description of vegetation classes.





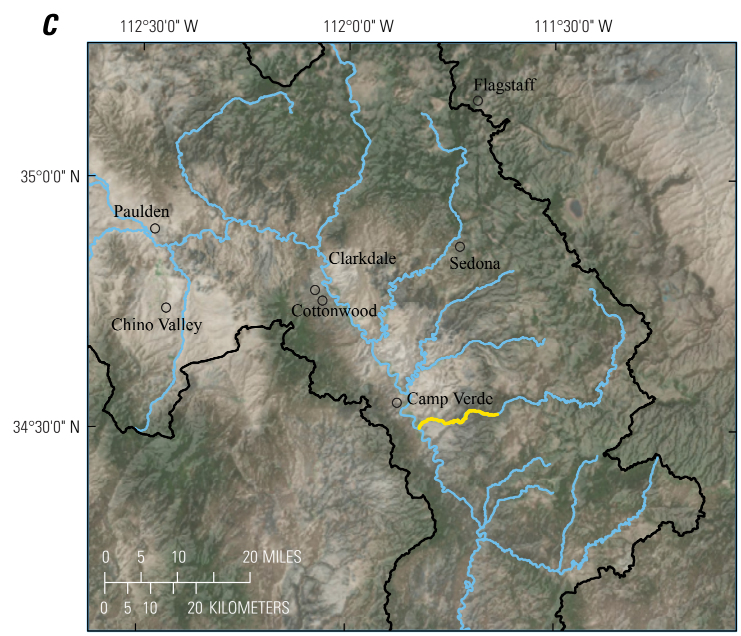


Figure 19. Graph and pie chart showing distribution of vegetation along West Clear Creek, Arizona, from confluence with Verde River to sampling site 11 miles upstream. *A*, Graph of vegetation-class area per river mile; *B*, pie chart of percent break-down of vegetation types. *C*, Study area map showing river reach in yellow (black line, watershed boundary; see fig. 1 for map location). See table 6 for description of vegetation classes.

the images provided by Yavapai County Flood Control District and Bing Maps as linked to the ArcGIS base map. Geomorphic channel features were identified by visual characteristics, such as whether the water surface was rough or smooth, the channel width and sinuosity, and obstructing features that might cause a backwater. Pools were identified by smooth water surfaces and wider channel widths and were typically found where a downstream obstruction created a backwater. Runs were identified as generally smooth water surfaces with narrower stream widths than pools. Riffles were identified by rough water surfaces with steeper gradients than pools and runs and were typically found where the river flowed over gravel bars (Konrad and others, 2008a).

Figures 20 and 21 show the distribution of geomorphic units throughout the middle Verde River; the upper-middle reach had more riffle habitat and less pool habitat than the lower-middle reach. The tributary Oak Creek (fig. 22) and West Clear Creek reaches (fig. 23) were also digitized. The Oak Creek reach was more homogenous in geomorphic stream-habitat composition than the West Clear Creek reach.

Runs dominated in the upper sections of West Clear Creek, and pools were more dominant in many of the lower sections; riffle habitat was more common in the upper sections than in lower sections. This difference between the two reaches was likely related to the greater slope and sinuosity of the West Clear Creek channel and floodplain. In some areas such as lower West Clear Creek, canopy cover can occasionally block channel visibility on aerial imagery, and the accuracy of the fluvial geomorphic descriptor classification can be reduced. As with the analysis of the distribution of riparian vegetation, this distribution of geomorphic units can be used as a basis for analysis of in-stream habitat.

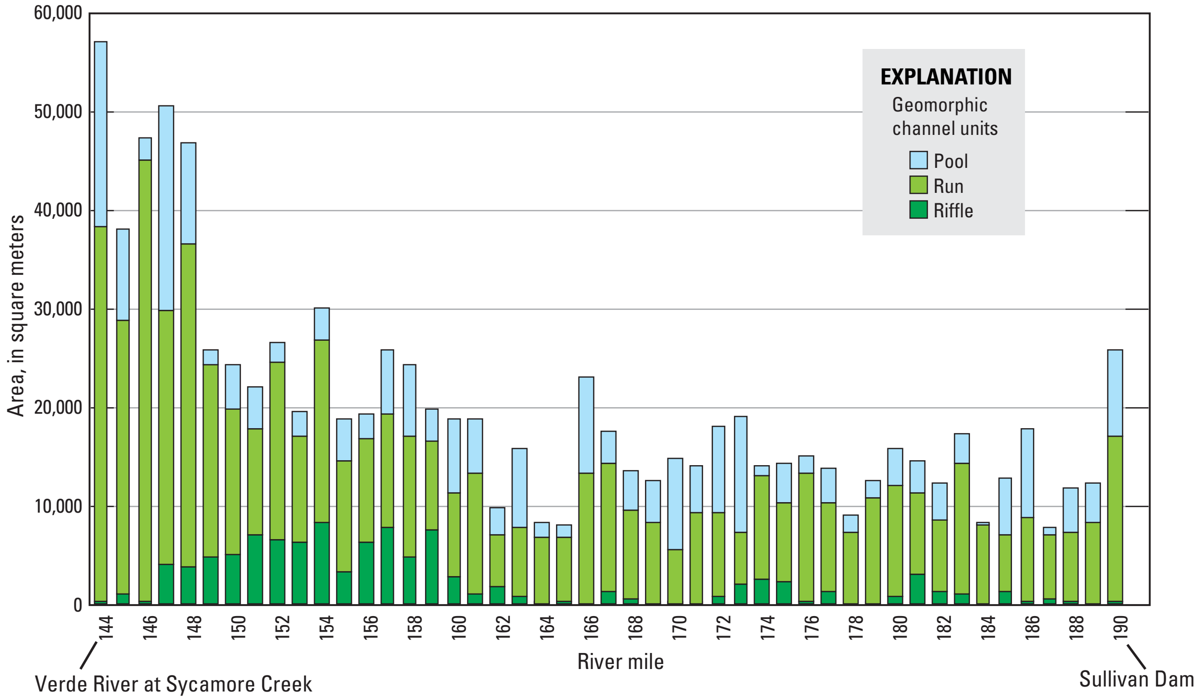


Figure 20. Graph showing upper-middle Verde River, Arizona, geomorphology class distribution per river mile.

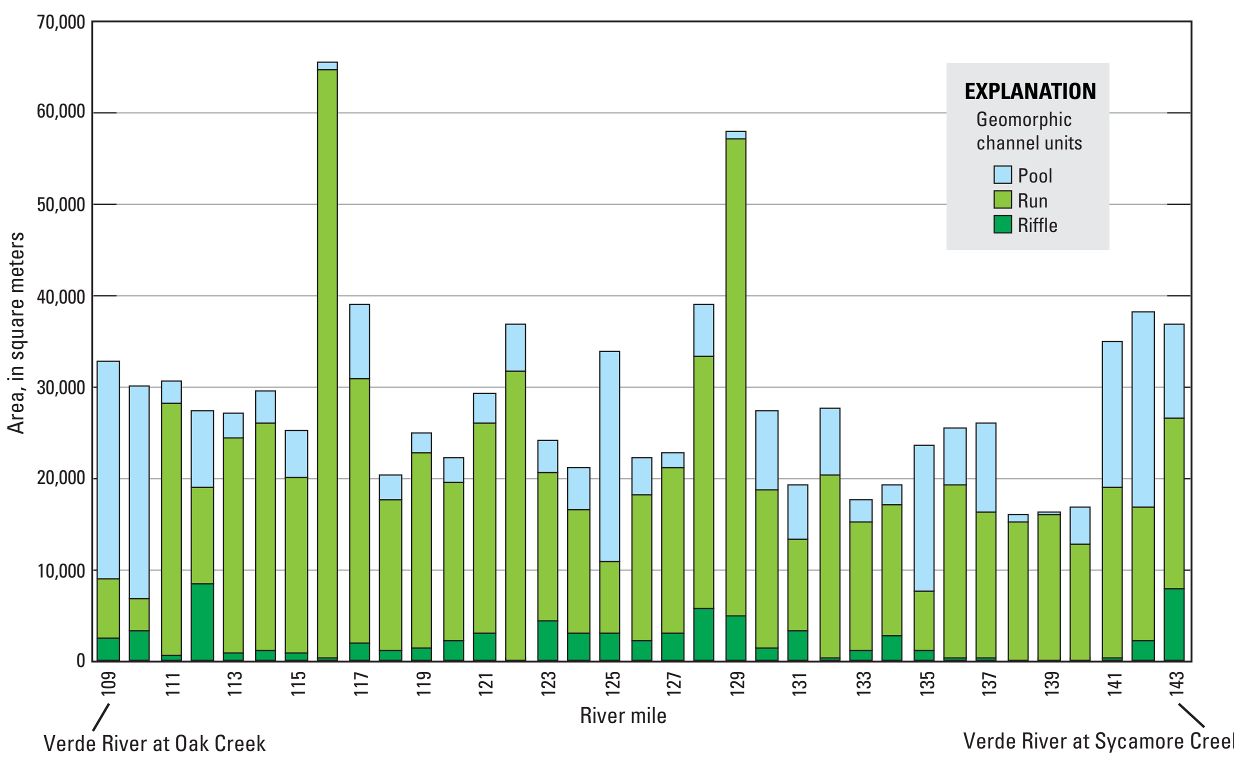


Figure 21. Graph showing lower-middle Verde River, Arizona, geomorphology class distribution per river mile.

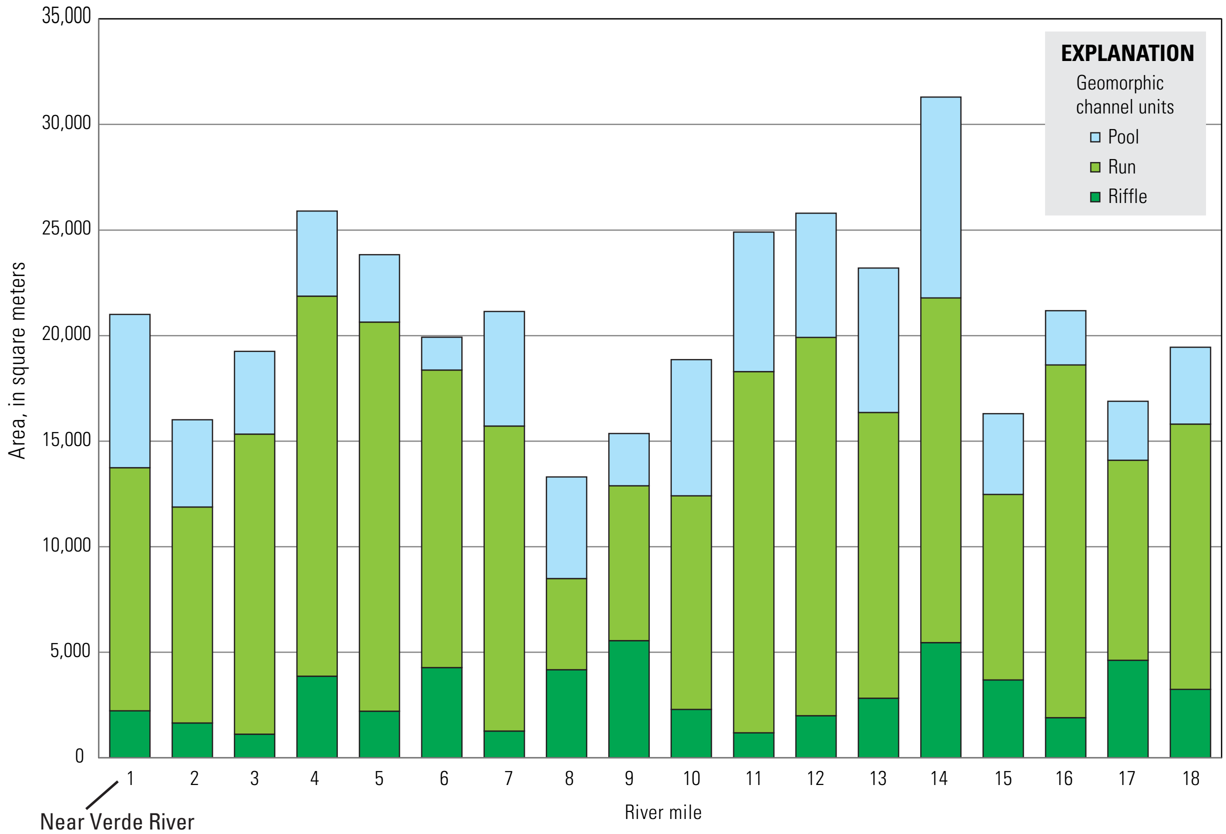


Figure 22. Graph showing Oak Creek, Arizona, geomorphology class distribution per river mile.

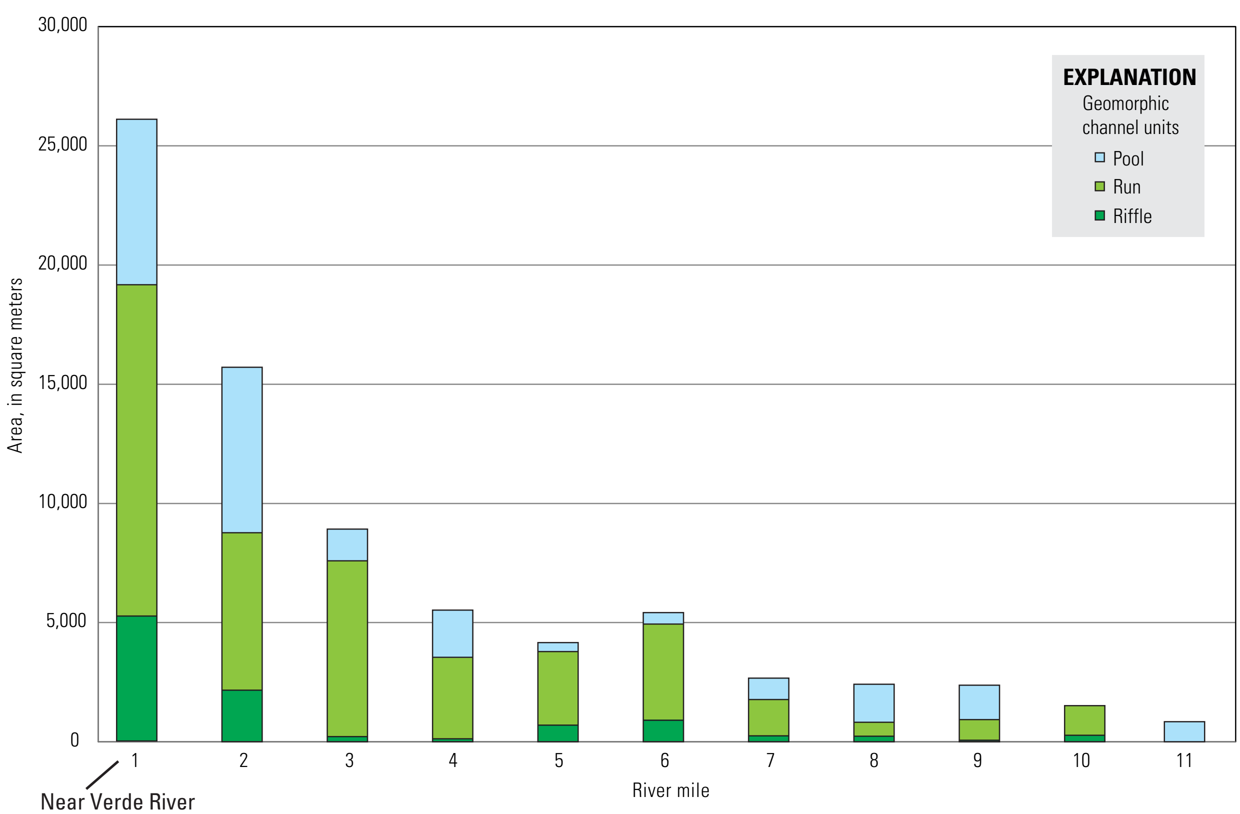


Figure 23. Graph showing West Clear Creek, Arizona, geomorphology class distribution per river mile.

Table 7. Geomorphic channel units used in this study of the middle Verde River, Arizona (Moulton and others, 2002).

Geomorphic channel unit	Description
Riffle	Turbulent flow; shallow, coarse-grained substrate.
Run	Laminar flow, less turbulent; variable depth and substrate.
Pool	Very low current velocity; relatively deep, depositional accumulations of fine sediment particles.

Bedload Transport at the Paulden Streamgage

Models of flow and sediment transport have been used to interpret the fluvial geomorphic processes that affect the mechanisms that influence habitat development, water quality (suspended sediment), and organism abundance and distribution (Wiele and others, 2007). As a first step toward such an understanding of the fluvial geomorphology of the Verde River, the usefulness of the Hydrologic Engineering Center River Analysis System (HEC-RAS) model sediment transport output for the Verde River was evaluated at different time steps (U.S. Army Corp of Engineers, 2018).

In general, sand-bedded channels can be readily reworked even at moderate and low discharges. At high flows, significant scour and fill can occur in the channel, and the sediment load can vary significantly as a function of discharge depending on sediment supply. In contrast, gravel-bed channels, which dominate the upper-middle and lower-middle Verde River, tend to be far less active and provide a more stable framework for in-stream organisms, although sand-sized sediment is still deposited in pools. Hydraulic properties tend to vary smoothly with water-discharge, but sediment transport, especially in gravel-bedded rivers such as the Verde, tends to be minimal until a threshold is reached. As water-discharge increases beyond this threshold, sediment transport increases sharply as a nonlinear function of bed stress. Available data were used to estimate the gravel-transport threshold and transport rate as a function of water-discharge at the Verde River near Paulden gage. Flow and channel properties and bedload flux at the Paulden gage were estimated from existing data in combination with data from field samples. Hydraulic properties were estimated from a peak-flow study conducted by the USGS, and bed-material size distribution was determined from recent pebble counts.

Given the abundance of bed material and the tendency toward infrequent and short duration high flows, a relation between water-discharge and bedload derived from sediment transport can be used to estimate the history of sediment transport at the Paulden gage. A similar relation could not be used for fine sediment in the Verde River because transport of fine sediment is likely constrained by supply. Discharge records at 15-minute intervals are available from 1987 onwards from the USGS National Water Information System (NWIS; https://waterdata.usgs.gov/nwis). To directly compare how channel properties, flow properties, and sediment flux varied with water-discharge, these values were nondimensionalized so that all were 0 at the lowest discharge computed and 1 at the highest discharge computed (fig 24*A*). The sediment flux was relatively dormant until the water-discharge reached a nondimensional threshold value of about 0.2, beyond which sharply elevated sediment transport occurred. This nondimensional water-discharge value (0.2) corresponded to the real sediment-discharge of 0.01 cubic feet per second (ft^3/s) shown in figure 24*B*.

The sediment transport computed for 22 discharges modeled with the HEC-RAS computer program were used as a linearly interpolated table to compute sediment transport over the 15-minute gage hydrographs. Mobilization of bed material has been an infrequent and irregularly timed event at the Paulden gage since 1987. Five events exceeded the elevated sediment (or gravel) transport threshold (sediment-discharge of 0.01 ft^3/s) from water year 1987 through water year 2011 (fig. 24*B*, 15-minute data). A gap of 11 years occurred between the 1993 and 2004 events in which channel-bed material was not disturbed; that period ended with two events exceeding the gravel transport threshold only 5 months apart. The poor sorting and angularity of bed sediment near the Paulden gage is consistent with infrequent transport.

The sensitivity of sediment transport to discharge makes the temporal resolution of discharge records worth considering. Fifteen-minute resolution is generally sufficient to capture the rise, peak, and fall of runoff events even in the hydrologically volatile arid Southwest. However, 15-minute data can be cumbersome to work with, may not be available, and may not be compatible with some software. On the other hand, using average daily discharges (red plot in fig. 24*B*) to compute bedload transport at the Paulden gage yielded just 3 of the 5 events evident in the 15-minute calculations (blue line in fig. 24*B*), which yields much lower estimates of transport rate. Monthly average discharge data yielded no months that exceeded the gravel-transport threshold; neither can monthly records of streamflow capture the channel-forming processes that are a fundamental part of habitat development and maintenance. Daily flows are a significant improvement from monthly averages, but considering the volatility of storm runoff in the Verde River watershed, the impact of some events could be obscured by averaging discharge over daily time steps. When possible, 15-minute data should be used to evaluate sediment transport in the Verde River system.

Use of cross sections and roughness estimates obtained following the 1993 peak of record (22,300 ft^3/s) degrade the accuracy of the calculations over the 24-year discharge record because the peak flow altered the channel, and channel properties have changed over time. Vegetation was removed during the 1993 flood, so the channel was likely smoother than at other times, and the channel configuration has likely been altered from its previous shape by the 1993 flows and by the subsequent high flows. However, for ecological considerations, the estimates of frequency, intensity, and duration of channel-forming events are still useful.

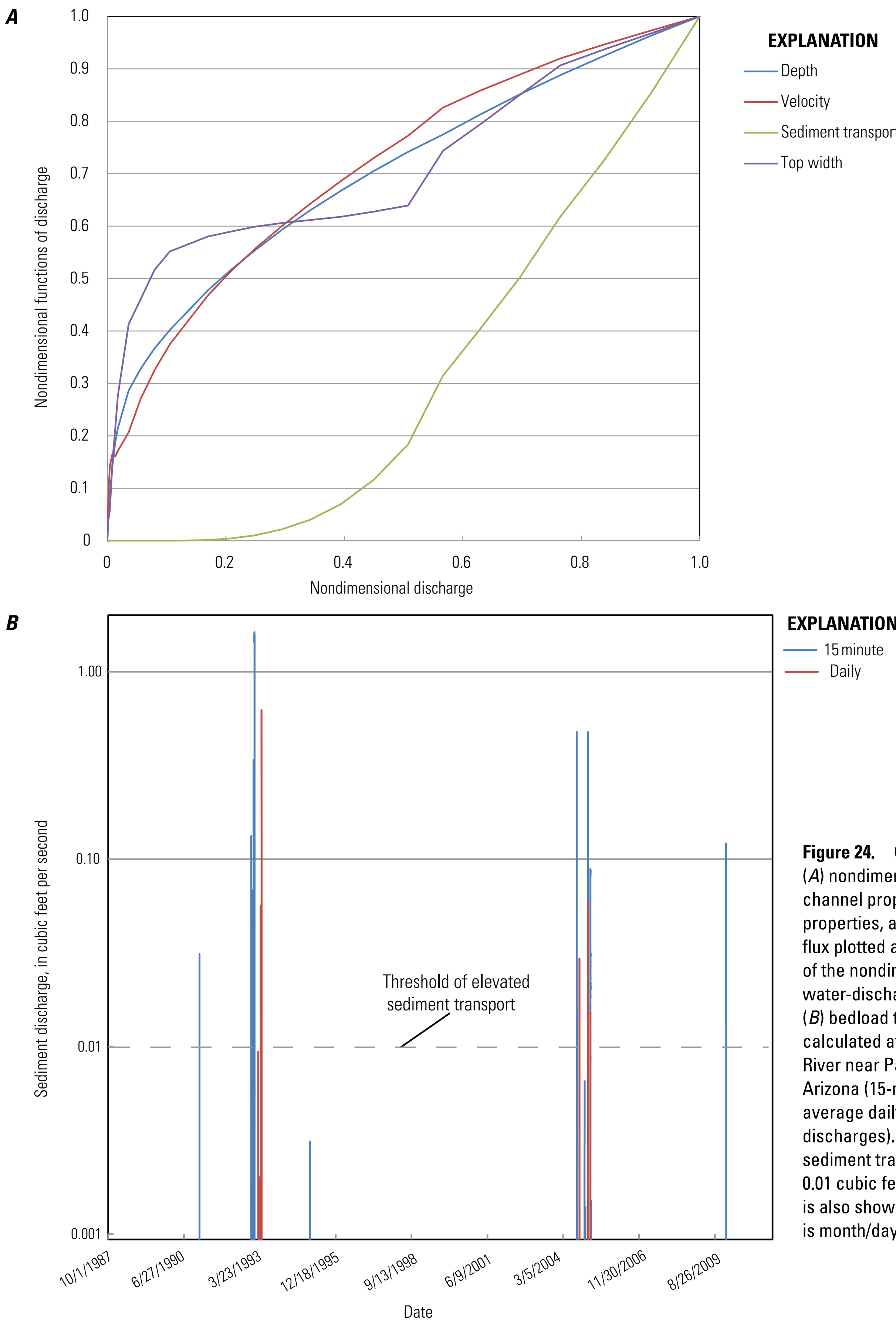


Figure 24. Graphs showing (*A*) nondimensionalized channel properties, flow properties, and sediment flux plotted as a function of the nondimensionalized water-discharge and (*B*) bedload transport calculated at the Verde River near Paulden gage, Arizona (15-minute and average daily sediment discharges). The significant sediment transport line of 0.01 cubic feet per second is also shown. Date format is month/day/year.

Fish and Macroinvertebrates

Analysis Framework

The middle Verde River watershed is a unique and important resource that provides large areas of undeveloped habitat to a diverse range of animal species. The middle Verde River is one of the few remaining free-flowing reaches found on large rivers in Arizona and is recognized by local, State, and Federal environmental- and natural-resource agencies as a sensitive and vulnerable resource. Although the USGS, U.S. Environmental Protection Agency (EPA), United States Fish and Wildlife Service, USFS, Arizona Game and Fish Department (AZGFD), and Arizona Department of Environmental Quality (ADEQ) have all conducted research and monitoring in the Verde River watersheds, not all information is compatible for compilation because of differences in objectives, time, space, and scale. Research to understand how non-native species and water quality have been changing over time has been ongoing for many decades, but much less work has been completed that directly relates the hydrologic and flow alteration characteristics of the main Verde River and tributaries to the function of ecological communities. These different agency data sources and data types were compiled here to be used in a preliminary analysis of fish and macroinvertebrate communities in relation to hydrologic and habitat characteristics in the middle Verde River watershed. The different data sources required the consideration of factors related to the timing of collection (the year and season), the location of collection with respect to scale (watershed, reach, and microhabitat), and streamflow conditions before collection (timing and magnitude of recent flooding events).

In addition to analyzing existing data, a fish pilot study and a macroinvertebrate pilot study were done. The intent of these pilot studies was to provide information about the design and implementation of ecological flow studies and about how a study could be scaled up to investigate the ecological and hydrological relations between biotic assemblages and habitat data. The pilot studies were (1) a collaboration between Northern Arizona University (NAU) and the USGS for the purpose of better understanding the eco-hydrology relations of fish communities at the reach scale and (2) a USGS macroinvertebrate and habitat analysis in the upper-middle Verde River.

Fish and Macroinvertebrate Data Sources and Processing

There were 312 fish surveys conducted in the Middle Verde River watershed between 1992 and 2011 and of the 312 surveys, 292 were conducted along the Verde River from below Stillman Lake (near the confluence with Granite Creek) to the stream gaging station Verde River near Camp Verde (09506000; fig. 25). The other 20 surveys were conducted at West Clear Creek, Wet Beaver Creek, and Oak Creek. Macroinvertebrate samples used in the analysis were all collected between 1992 and 2010 and came exclusively from riffle habitat (generally the faunistically richest in-stream habitat type) along the upper-middle and lower-middle Verde River and many of its tributaries. Of the 183 riffle samples analyzed, 27 were from the upper-middle Verde River, 19 from the lower-middle Verde River, 11 from Sycamore Creek, 5 from Bitter Creek, 47 from Oak Creek, 8 from Spring Creek, 29 from Wet Beaver Creek, and 37 from West Clear Creek (fig. 26).

Most fish surveys that have been completed along the Verde River were conducted by the AZGFD, and methods can be found at http://www.azgfd.gov/w_c/tech_reports.shtml. A part of the Verde River and all of the West Clear Creek fish surveys were collected by the USGS using EPA methods (Peck and others, 2006) or USGS National Water Quality Assessment (NAWQA) program protocols (Moulton and others, 2002). Sampling methods consisted of either backpack or canoe electrofishing, and species identification and counts were completed in the field. The assemblage data collected were specific to each agency's objectives and were not specifically collected for the intended purpose of an environmental-flows investigation.

Existing macroinvertebrate data were compiled and compared for a combined analysis. Macroinvertebrate samples were collected using ADEQ methods (Arizona Department of Environmental Quality, 2006), EPA Environmental Monitoring and Assessment Program (EMAP) methods (Peck and others, 2006), or NAWQA program protocols (Moulton and others, 2002). A detailed comparison between ADEQ and EMAP methods can be found in Spindler and Paretti (2009). Samples were collected using a net with a 500-micrometer mesh from riffles within a given stream reach (coarse substrate), which generally provide habitat for the faunistically richest communities (Moulton and others, 2002). Samples were collected within a defined area to standardize the sampling effort and make it possible to compare sites. Samples were identified by Aquatic Biology Associates, EcoAnalysts, or the USGS National Water Quality lab to species or genus level using a fixed-count of 300 or 500 organisms.

The macroinvertebrate data were combined and processed through the software package Invertebrate Data Analyses System to resolve the occurrences of ambiguous taxa (Cuffney, 2003). Ambiguous taxa are those taxa whose identifications cannot accurately be determined to the lowest common taxonomic level. Commonly this is a result of either damaged or immature individuals. For this study, taxa were only identified to order to maintain consistency between various agency data.

Statistical Analysis and Methods

Several descriptive, univariate, and multivariate statistical approaches were applied to describe and summarize biotic metrics and assemblages and to develop abiotic associations with assemblage data. In the following sections, descriptive statistics are presented mostly as visual distributions in the form of boxplots and as summary statistics in the form

Figure 25. Map showing State of Arizona and Federal agency fish survey locations on the upper-middle and lower-middle Verde River, Arizona, and tributaries color coded by percentage of native fish present.

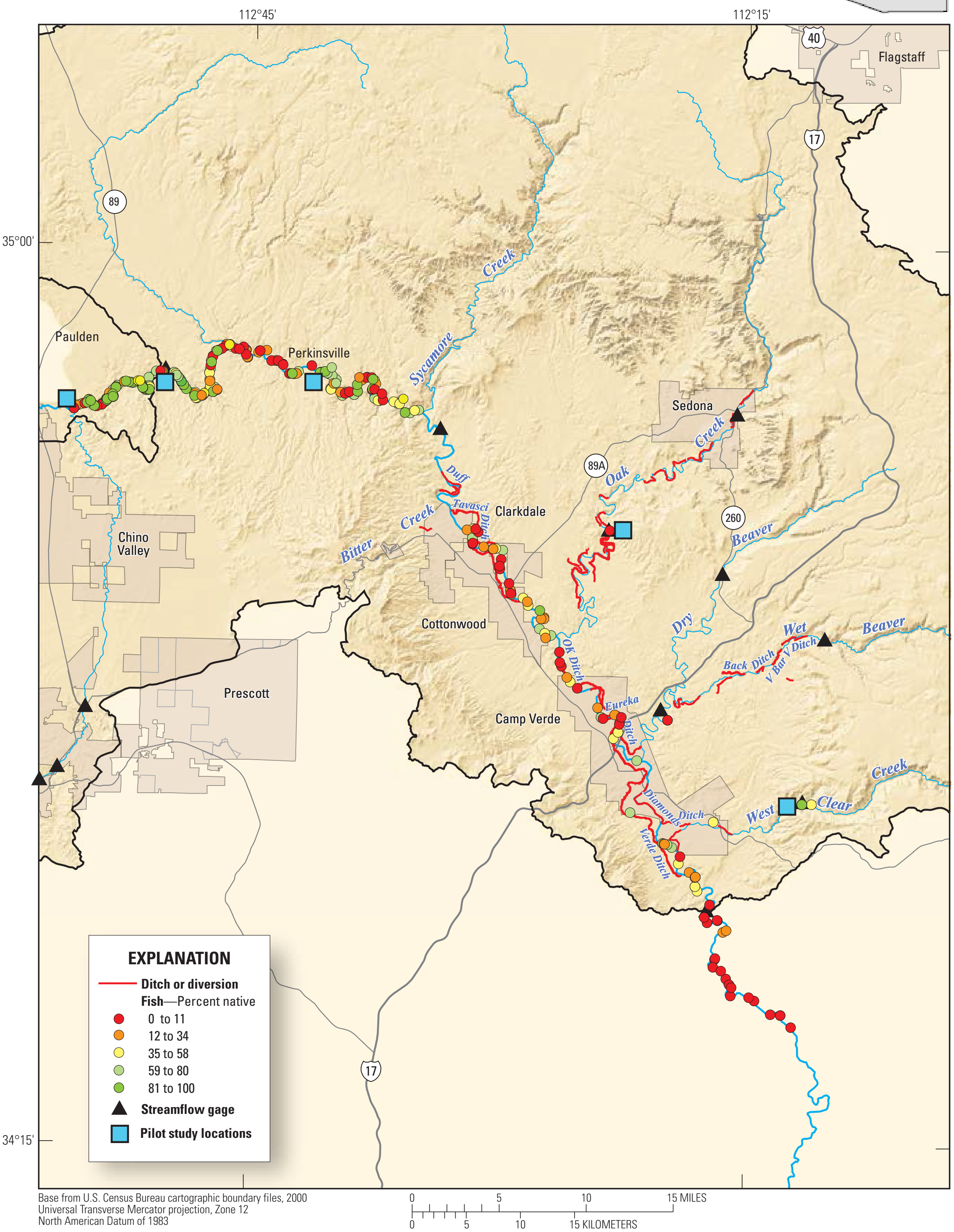
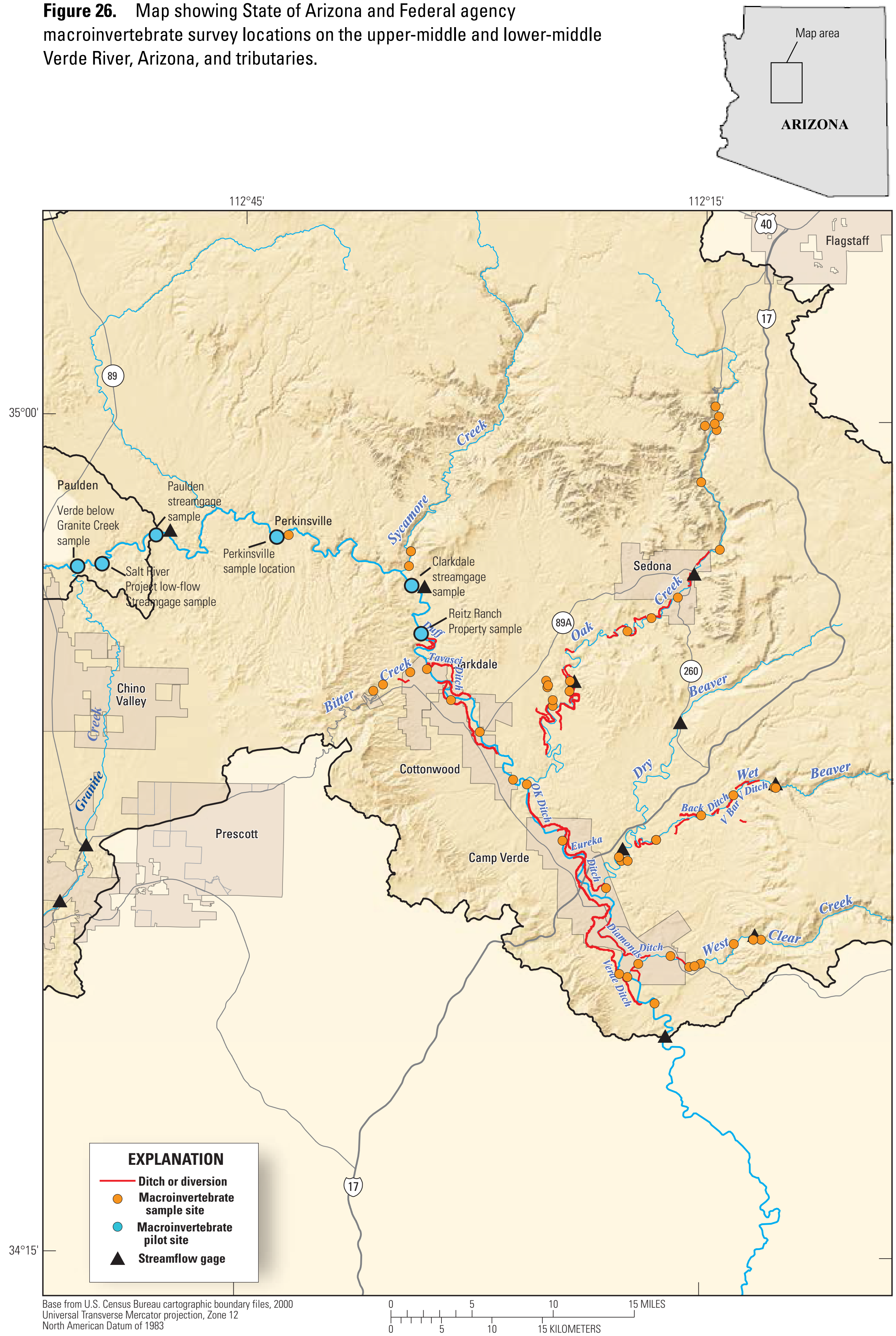


Figure 26. Map showing State of Arizona and Federal agency macroinvertebrate survey locations on the upper-middle and lower-middle Verde River, Arizona, and tributaries.



Base from U.S. Census Bureau cartographic boundary files, 2000
Universal Transverse Mercator projection, Zone 12
North American Datum of 1983

of mean, median, percentiles, and standard deviation. Nonparametric statistics were used to compare groups and trends, again to avoid normal data-distribution assumptions required for parametric statistics. Pairwise Wilcoxon rank-sums test was used for statistical comparisons between groups; monotonic relations between flow metrics and biota were analyzed using a nonparametric Kendall's tau (τ)correlation procedure, which again makes no assumptions about the normality of the data distribution. Nonparametric regression procedures, such as spline fitting and locally weighted scatterplot smoothing, were used to understand trends and biotic responses to abiotic variables such as hydrologic metrics and stream habitat measures.

Assemblage data were examined using a robust nonmetric multidimensional scaling (NMDS) multivariate analysis procedure. NMDS was used to examine patterns in assemblage datasets. The NMDS analysis was performed on a Bray-Curtis similarity matrix and plotted in two-dimensional space to aid in interpretation of multivariate patterns. A measure of stress is provided from the NMDS analysis as a diagnostic to determine how well the data are fitting the NMDS ordination. Lower stress values are desired, and stress values below 0.20 generally indicate that the NMDS plot is providing an accurate representation of the data in multivariate space. Many times, a multivariate centroid analysis was used to further simplify the visualization of the NMDS structure. This procedure essentially sums the distances of each observation (sample assemblage) relative to the observations within a group (within group sum of squares) and compares the distances to all other observations in all other groups (total sum of squares), and the measure of central location for the group constitutes a centroid location for that group, which can then be used in group comparisons.

Vector overlays were added to NMDS graphical outputs as an additional exploratory tool to visualize potential linear or monotonic relations between influential taxa and the ordination axes. Spearman rank correlations were restricted to a vector length of ±0.50 or greater, with ±1.0 indicating complete correlation. The length and direction of each vector indicates the strength and sign, respectively, of the relation between the taxa and the NMDS axes (Anderson and others, 2008). Similar exploratory visualizations were presented as bubble plots, where habitat and flow variables were superimposed onto the NMDS plot to better visualize patterns between the community and influential habitat features. Each bubble represents the site assemblage, and the size of the bubble represents the magnitude of the habitat variable measured at the site. The purpose for this procedure is to visualize patterns and associations between abiotic measures and assemblage data that may not be obvious in bivariate space.

A subset of fish and macroinvertebrate metrics that were significantly correlated ($p<0.05$) to hydrological metrics were used in quantile regression analysis. Fish and macroinvertebrate datasets commonly display a "wedge shaped" response to an explanatory habitat variable, indicating unequal rates of changes at different quantiles and a threshold or limit of the response to that variable (Terrell and others, 1996; Konrad and others, 2008b). Quantile regression fits a continuous function through the local (with respect to the independent variable) value of the quantile of a dependent variable to account for variation in the quantile with the independent variable. The method quantifies the rate of change in the dependent variable, which includes the upper and lower ends, whereas a least-squares regression analysis quantifies the mean response in relation to the explanatory variable (Cade and others, 1999). In this analysis, the quantiles (0.1, 0.25, 0.50, 0.75, and 0.90) of the response variable (biotic metric) were fitted as linear functions of the explanatory variable (hydrological metric), similar to the method described in Kail and others (2012). The regression line with the largest quantile with the narrowest 90-percent confidence intervals for the regression-line slope that did not include zero was selected as the limiting threshold in the biotic response (Cade and others, 1999). The quantiles were tested for statistical significance in terms of the probability that the slope and intercept were zero.

Description of Hydrologic Metric Determination and Flooding Classification

Discharge records from the gaging stations on the Verde River and its tributaries were used to examine the relations between biota properties and streamflow. Streamflow properties that are potentially significant to biotic communities are represented by metrics that are measures of flow magnitude, duration, variability, and rates of change. For this study, sampling sites within 10 km of a gaging station were chosen for comparing biotic and streamflow metrics.

The hydrologic metrics computed by EFASC (Konrad, 2011; table 8) were used in the analyses. The metrics that represent biotic properties may be functions of antecedent flow conditions; however, the most significant time periods are unknown before analysis and may vary among taxa. Consequently, metrics were calculated over a range of time periods preceding and concurrent with the sampling dates. Metrics were calculated for 10-, 30-, 90-, 365-, and 1,095-day periods before the sampling dates if the discharge record was largely complete over that time period. However, in the end, only 30-, 90-, and 365-day periods before the sampling dates were used; the metrics computed by EFASC were not meaningful for the shortest and longest time periods.

EFASC was intended for the analysis of long-term streamgage records but can be used for selected time periods extracted from longer records (Konrad, 2011). In this application, EFASC is first applied to the period of record to compute streamflow threshold values that are specified as input for applications to shorter time periods. Flow metrics represent the ecologically significant characteristics—magnitude, timing, frequency, duration, and rate of change (Richter and others, 1996; Poff and others, 1997).

Large floods have been shown to be beneficial for the succession of fish species native to the Southwest (Rinne

2005; Propst and Gido (2004), and biota that have evolved under conditions of such variable discharge will return to a proper functioning condition following a flood. The magnitude and timing of a large flood are important to know because they affect the characteristics of the succession period that follows. Because the timing of sample collection is an important factor in ecological studies, individual hydrographs were evaluated to determine recent flow conditions for each year respective of when surveys and samples were collected. Flooding will typically mobilize woody debris and bed substrate, evacuating fish, macroinvertebrates, and algae, effectively resetting the river ecosystem and biotic communities. The establishment of a new community will vary depending on the flood characteristics, as well as the location and season. It is important to consider these factors when conducting a community analysis.

Samples collected in a given year were classified into three major flood-event categories—low, moderate, and high-flood years. The classification was modified from Minckley and Meffe (1987) who observed significant flooding effects, specifically species removal, on non-native populations at two orders of magnitude greater flow than the average discharge for a given station. Samples collected in years with flows that exceeded two orders of magnitude greater than the average discharge were identified as high-flood years, samples collected in years with flows that exceeded one order of magnitude greater than the average discharge were classified moderate-flood years, and samples collected in years with flows less than one order of magnitude greater than the average discharge were classified as low-flood years. Research pertaining to the characteristics of large flood events and beneficial effects to aquatic biota is scarce. Research is especially lacking in the quantification of the magnitude, duration, timing, and number of peaks necessary to promote native species success. The classifications presented in this investigation are meant to serve as a way to help distinguish the effects of natural events from other factors related to changes in biotic communities (fig. 27).

Reach Classification

To assess the difference in effect of physical variability from human impacts, two approaches were used in the analysis of existing fish and macroinvertebrate data—(1) sample locations classified into separate reaches based on the physical basin characteristics of the river (fig. 28) and (2) sample locations classified into separate reaches based on human alteration of river flow (fig. 29). River reach characterization is a statistical process in which segments of stream miles are differentiated by physical, climatic, and hydrological characteristics. This classification process provides a physical standard condition for comparing the aquatic biota observed in each classified reach. First characterizing the stream reaches by physical conditions provides an initial framework for identifying background ecological conditions so that subsequent relations between ecological metrics and flow alteration can be developed (Arthington and others, 2006). For example, the physiography and climate of the upper-middle Verde River near the confluence with Granite Creek is very different from the lower-middle Verde River near the confluence with West Clear Creek, and this spatial variability is reflected in the biotic communities that inhabit the different stream reaches. Two important ecosystem drivers—mean annual precipitation and mean elevation—can vary as much as 10 inches and 3,000 ft within the study area, respectively.

Reaches were differentiated based on differences in the physio-climatic characteristics in order to better understand the ecological response to measures of hydrologic changes or alteration throughout the study area. The USGS StreamStats program was used to compute basin characteristic statistics for each fish and macroinvertebrate sampling location. A preliminary correlation and scatter plot analysis was used to reduce the large number of 50 basin characteristic variables to a table of nonredundant variables that could be used to classify reaches. Highly correlated variables were removed using a Kendall's tau correlation coefficient scatter plot ($\tau \geq 0.7$). Next a multivariate principal component analysis with a varimax rotation was used to identify six nonredundant basin characteristics that differentiated reaches along a principal component gradient. The first and second principle components explained 97 percent of the variation in the dataset. The basin characteristics influential along this component axis included watershed stream density, mean annual precipitation, percent forest, percent upland herbaceous shrubs, soil permeability, and soil water capacity. The reaches were defined using these variables, and breakpoints were usually determined where a major contributing tributary entered the upper-middle and lower-middle Verde River (fig. 28).

Multivariate centroids were determined for a matrix of the six variables and plotted using NMDS. In an NMDS plot, centroids that plot closer together are more similar. The NMDS plot (fig. 28, inset) thus showed that reaches 1 and 2 were more similar because both are within the upper-middle Verde River where similar geomorphology and climate distinguish these reaches from downstream locations of the lower-middle Verde River. The upper-middle Verde River has more small, contributing tributaries with more herbaceous uplands and less sandy soils than the lower-middle Verde River. The water capacity and storage of the upper Verde River is therefore greater too. Lower in the watershed (reaches 4 and 5) water is draining from the higher elevation forested areas of the watershed. The lower-middle Verde River (beginning with reach 3) widens markedly from the upper-middle Verde River, and basin characteristics change greatly. These basin characteristics affect surface-water runoff, storage capacity, and water chemistry which in turn affect the biotic communities using those river reaches. A new reach was generally defined at the watershed entry point of each subsequent major tributary. For example, reach 3 extends from Sycamore Creek (near the Clarkdale streamgage (09504000)) to the confluence with Oak Creek, reach 4 is between Oak Creek and West Clear Creek, and reach 5 is between West Clear Creek and the Camp Verde streamgage (09506000) (fig. 28). Reaches 3 and 4 cover most of the

Table 8. Streamflow metrics computed by the Environmental Flow Allocation and Statistics Calculator (Konrad, 2011).

Metric name	Description
Mean streamflow	Mean streamflow for the period of analysis: $(\Sigma_{d=1 \text{ to } D} Q_d)/D$, where d is each day and D is the number of days in the period of analysis.
Q_{01}	Streamflow equaled or exceeded 1 percent of the period of analysis
Q_{10}	Streamflow equaled or exceeded 10 percent of the period of analysis
Q_{25}	Streamflow equaled or exceeded 25 percent of the period of analysis
Q_{50}	Streamflow equaled or exceeded 50 percent of the period of analysis
Q_{75}	Streamflow equaled or exceeded 75 percent of the period of analysis
Q_{90}	Streamflow equaled or exceeded 90 percent of the period of analysis
Q_{99}	Streamflow equaled or exceeded 99 percent of the period of analysis
50th percentile of absolute value of percent daily change	The median absolute value of percent daily change in streamflow. Percent daily change is calculated for each day (d); $((Q_{\text{day1}}-Q_{\text{day0}})/Q_{\text{day0}}$. No-flow days are not included in the calculation. For odd-numbered series, the median is determined by ranking all values for day 1 to day d in a series from highest to lowest and selecting the value at the d/2+1 place. For even-numbered series, the value is the midpoint between the integer part of d/2+1 and the next (smaller) value.
Coefficient of variation of daily flows	Standard deviation of daily streamflow divided by mean streamflow.
Coefficient of variation of log-transformed daily flows	Coefficient of variation of log-transformed streamflow: $(10^{(Var(\log(Q))}-1)^{0.5}$, where $Var(\log(Q))$ is the variance of the series of log-transformed daily values. No-flow days are not included in the calculation.
Coefficient of variation of log-transformed annual maximum flow	Coefficient of variation of log-transformed streamflow: $(10^{(Var(\log(Q\max))}-1)^{0.5}$, where $Var(\log(Q_{\max}))$ is the variance of the annual (water year) series of log-transformed maximum daily values. Years when $Q_{\max}$=0 are not included in the calculation.
Median annual maximum streamflow	Median annual (water year) maximum daily streamflow.
$Q_{\max}/Q_{50}$	Median annual (water year) maximum daily streamflow ($Q_{\max}$) divided by median streamflow (Q_{50}).
Median annual number of high-flow events	Median of the annual (water year) number of periods when streamflow exceeds the high-flow threshold.
Median duration of the annual longest high-flow event	Median number of days of the annual (water year) longest period of time when streamflow exceeds the high-flow threshold.
Number of months with annual probability greater than 0.5 of having a high-flow event	Number of months (October, November, December, … , September) that had flows exceeding the high-flow threshold in more than one-half of the water years of the period of analysis. Values range from 0 to 12.
Mode of highest streamflow month	Most common month of maximum monthly mean streamflow. The time series of monthly mean streamflow is calculated for the period of analysis. The month with the highest monthly mean is determined for each year. A value of 1 indicates that January was most often the month with the highest mean streamflow for the year.
Median annual minimum streamflow	Median annual (low-flow year) minimum daily streamflow.
$Q_{\min}/Q_{50}$	Median annual (low-flow year) minimum daily streamflow ($Q_{\min}$) divided by median streamflow (Q_{50}).
Coefficient of variation of annual minimum streamflow	Standard deviation of the annual (water year) series of minimum daily streamflow divided by mean annual minimum daily streamflow.

Table 8.—Continued.

Metric name	Description
Median annual number of low-flow periods	Median of the annual (low-flow year) number of periods when streamflow is less than the low-flow threshold.
Median duration of the annual longest low-flow period	Median number of days of the annual (low-flow year) longest period of time when streamflow is less than the low-flow threshold.
Number of months with annual probability greater than 0.5 of having a low-flow period	Number of months (October, November, December, … , September) that had flows less than the low-flow threshold in more than one-half of the low-flow years of the period of analysis. Values range from 0 to 12.
Mode of lowest flow month	Most common month of minimum monthly mean streamflow. The time-series of monthly mean streamflow is calculated for the period of analysis. The month with the lowest monthly mean is determined for each year. A value of 1 indicates that January was most often the month with the lowest mean streamflow for the year.
Fraction of years with no-flow days	Fraction of years with days that had no flow.
Mean number of no-flow days in years with no-flow days	The total number of days with no flow averaged over years with no-flow days. Years with streamflow on all days are not included in the calculation. No-flow days are calculated for each lowflow year.
Coefficient of variation of annual mean streamflow	Standard deviation of mean annual (water year) streamflow divided by mean streamflow.
Storm-flow recession coefficient	The 10th percentile of percent daily change for days when streamflow was steady (no change) or receding. The 10th percentile represents a relatively fast recession rate indicative of storm-flow recession because the values are negative.
Base-flow recession coefficient	The median annual (water year) value of the median percent daily change for days when streamflow was steady (no change) or receding. The median represents a typical recession rate indicative of base-flow conditions.
Maximum monthly streamflow	The maximum of means for January, February, March, … , December.
Minimum monthly streamflow	The minimum of means for January, February, March, … , December.
Maximum coefficient of variation of monthly streamflow	The coefficient of variation of annual values of mean streamflow for each month (January, February, March, … , December) is calculated and the maximum value is used. The maximum value represents the month with the lowest predictability in streamflow magnitude.
Minimum coefficient of variation of monthly streamflow	The coefficient of variation of annual values of mean streamflow for each month (January, February, March, … , December) is calculated and the minimum value is used. The minimum value represents the month with the highest predictability in streamflow magnitude.
Normalized range of monthly streamflow	Median annual (water year) value of the difference between maximum monthly mean streamflow and minimum monthly mean streamflow divided by annual mean streamflow. If the annual mean streamflow is zero, then the normalized range of monthly streamflow is zero.
$\log(Q_{10}/Q_{90})$	Logarithm (base 10) of the streamflow exceeded 10 percent of the time divided by streamflow exceeded 90 percent of the time.

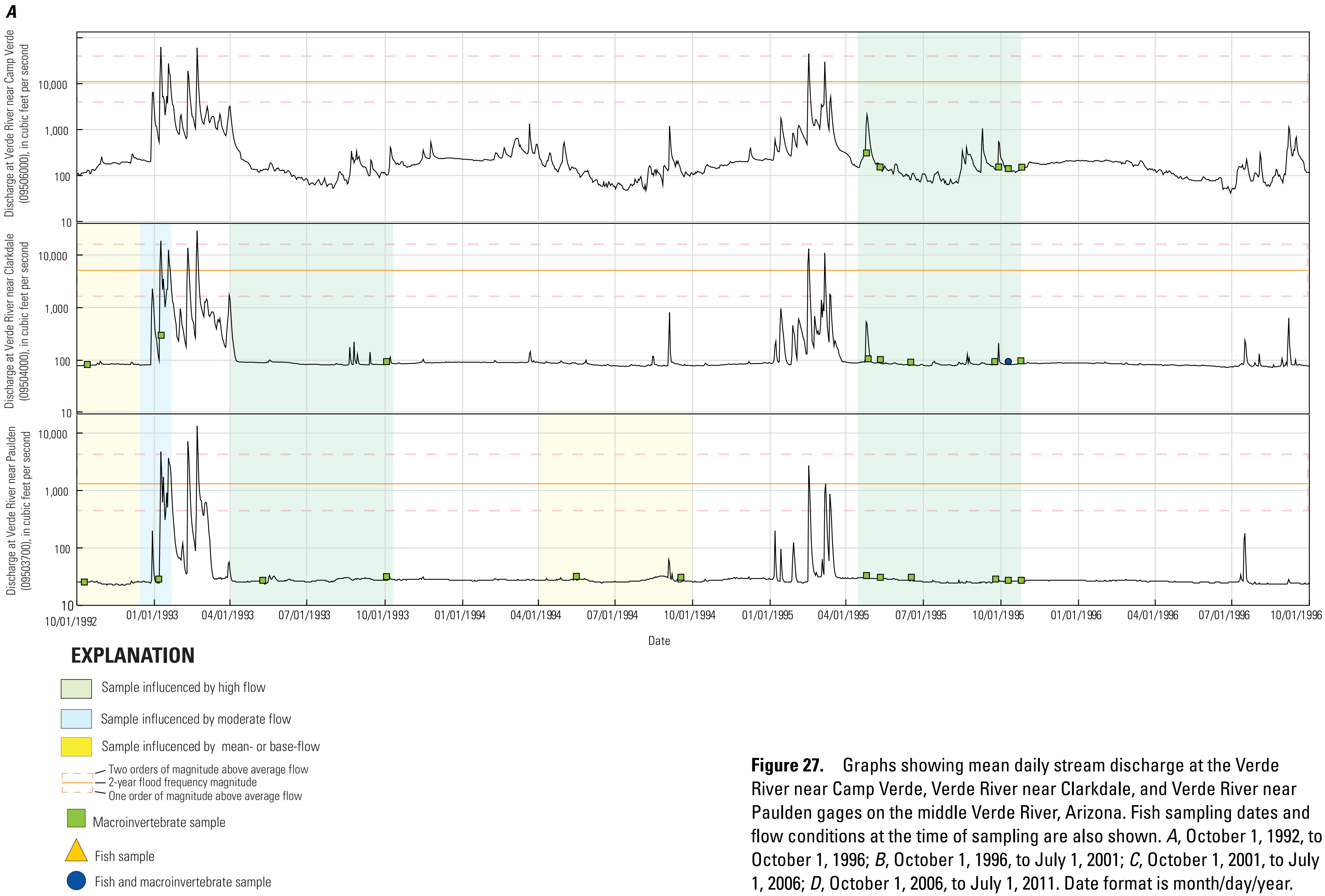


Figure 27. Graphs showing mean daily stream discharge at the Verde River near Camp Verde, Verde River near Clarkdale, and Verde River near Paulden gages on the middle Verde River, Arizona. Fish sampling dates and flow conditions at the time of sampling are also shown. *A*, October 1, 1992, to October 1, 1996; *B*, October 1, 1996, to July 1, 2001; *C*, October 1, 2001, to July 1, 2006; *D*, October 1, 2006, to July 1, 2011. Date format is month/day/year.

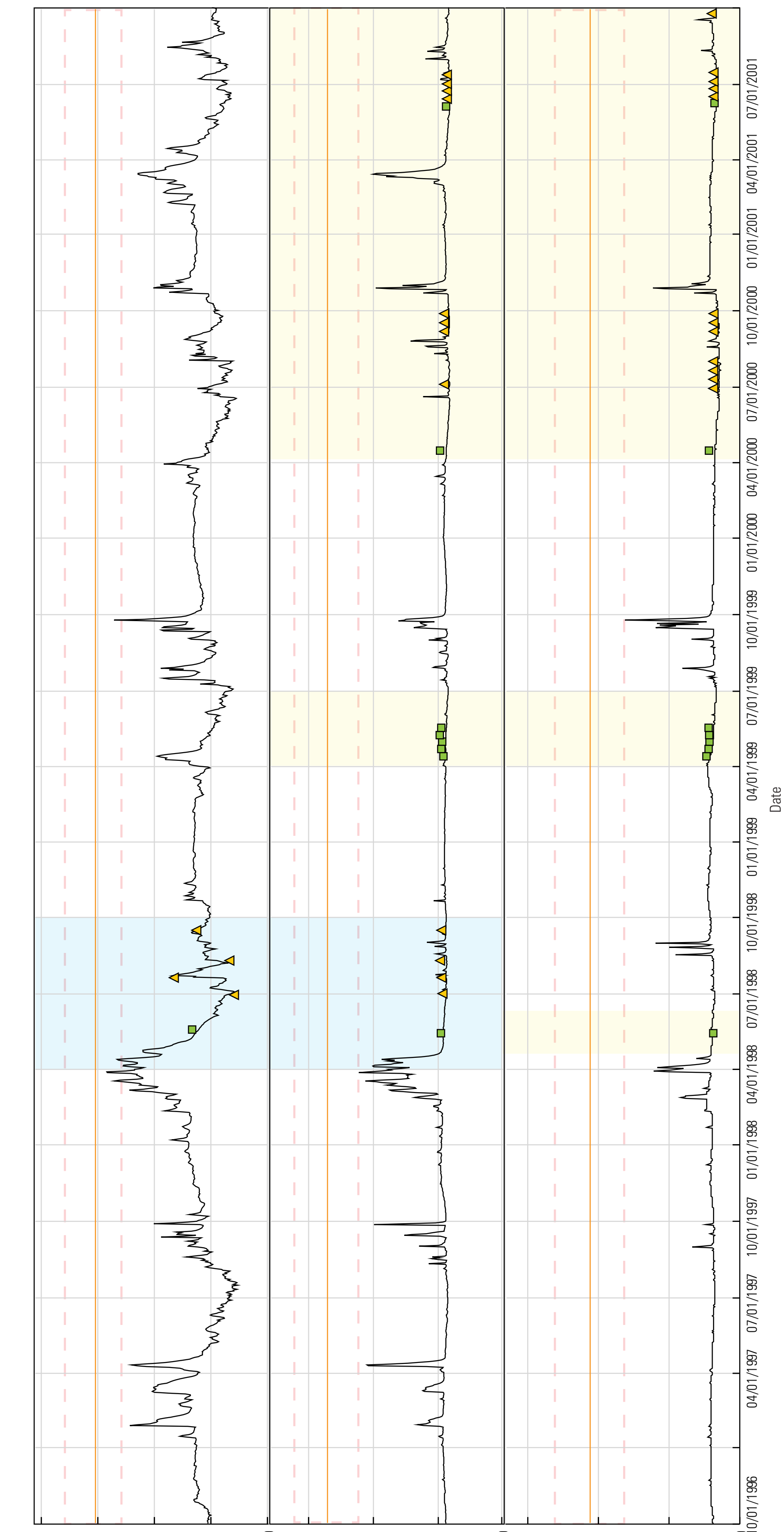


Figure 27.—Continued

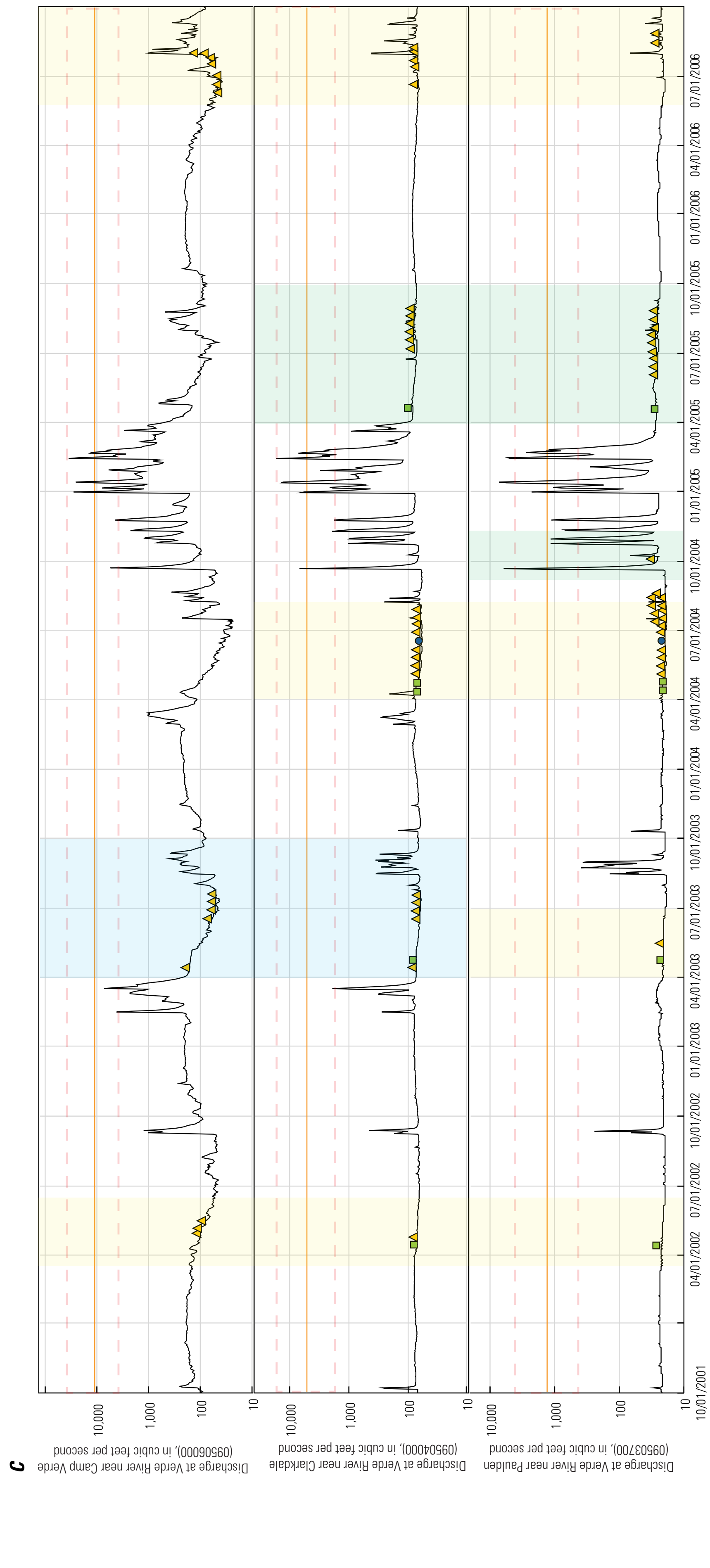


Figure 27.—Continued

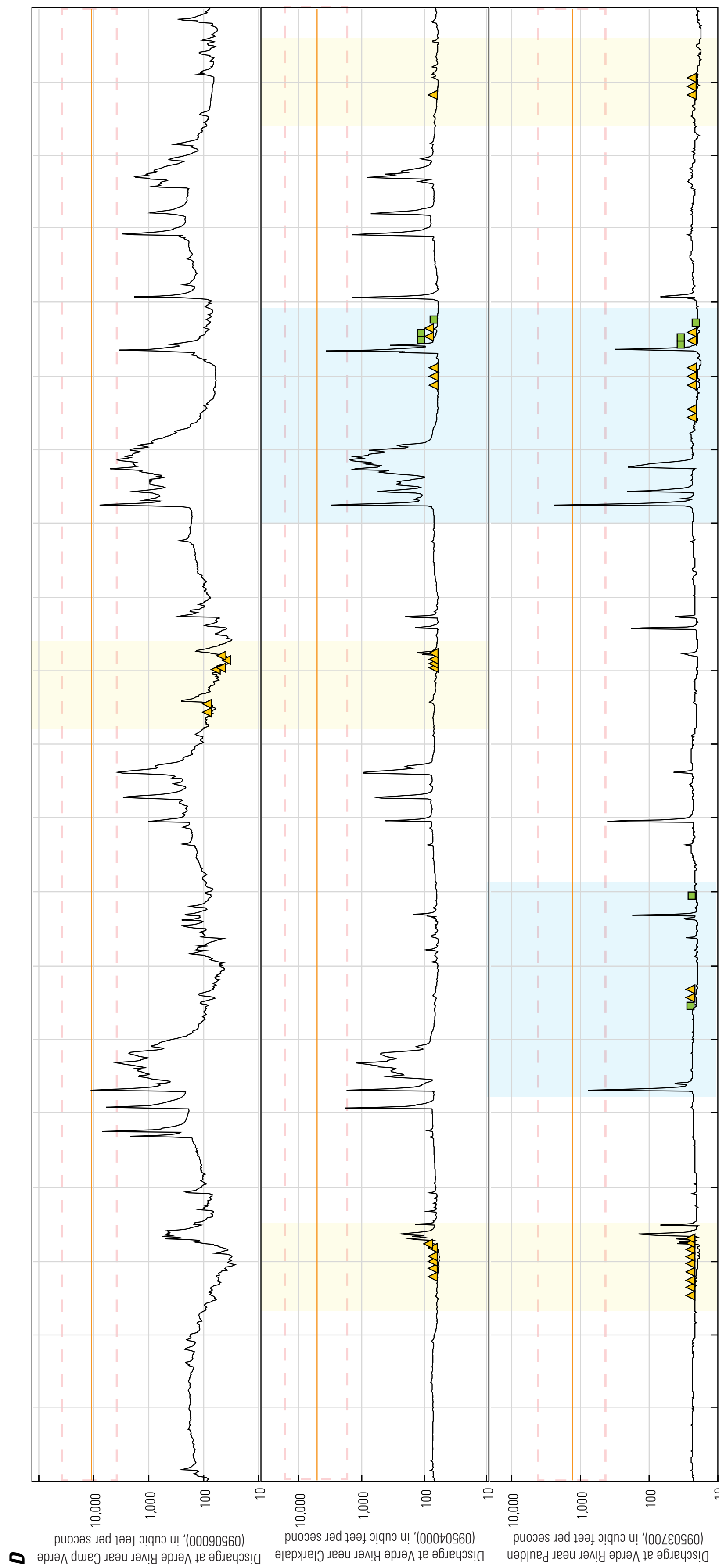


Figure 27.—Continued

significant diversions plus the urbanized areas of the study area, and the percent impervious surface is about 2 percent more than the other reaches.

The second reach-characterization approach was based on the type and amount of human alteration to reach flow. Following the classification scheme of Garner and Bills (2012), sampling locations were grouped based on known diversions in order to compare reaches of varying flow alteration (fig. 29). These reaches are indicated by Roman numerals as opposed to reaches defined by basin characteristics shown in figure 28.

Human activities such as groundwater pumping, surface-water diversions, flood control, and development in the floodplain can each cause alterations to the river's natural flow regime (Springer and Haney, 2008). Groundwater pumping eventually captures water that would otherwise discharge to a stream (Filippone and Leake, 2005; Leake and others, 2008; Garner and others, 2013), thus reducing the magnitude of base flow. Surface-water diversions also reduce the magnitude of base flow (Garner and others, 2013) and can increase the duration of extreme low flows. In some cases, diversion structures can result in dewatered reaches below the structure (Brasher, 2003). Flood-control structures and development in the floodplain may cause higher peak flows due to constriction of the flood plain and increased volume and velocity of runoff due to increased impervious cover. Springer and Haney (2008), Blasch and others (2006), Wirt and others (2005), and Alam (1997) all provide data, summaries, and discussion of groundwater pumping and surface-water diversion and their resulting impacts on the flow regime in the Verde River.

Synoptic base-flow surveys, or seepage runs, were conducted in the Verde Valley in June 2007 and February 2011 (Garner and Bills, 2012). Both surveys were conducted between the Clarkdale gaging station (09504000) and the Camp Verde gaging station (09506000), a distance of 51 river miles. These data were used to understand spatial variability in gaining and losing reaches and the effects that human alterations of the surface-water system have on base flow (Garner and Bills, 2012). The dozens of surface-water diversions, including several large gravity-fed ditch diversions, had an obvious influence on Verde River discharge. Results also showed that surface water contributed to the Verde River; the perennial tributaries of Oak Creek, Beaver Creek, and West Clear Creek were a major factor in explaining perennial flow in the Verde River.

The seepage-run investigation completed by Garner and Bills (2012) defined reaches in the middle Verde River watershed based on degree of surface-water diversions (fig. 29). Reach I–II begins below the Clarkdale gaging station and ends at the Tavasci Ditch and has no major surface-water divisions. Reach II–III includes the Tavasci, Hickey, and Cottonwood ditch diversions, and streamflow can be reduced by as much as 65 ft^3/s in summer months. Reach III–IV is about 6 river miles where there are no diversions, and streamflow increases. Reach IV–V is a highly diverted stretch where the OK, Eureka, Verde, and Diamond S ditches can divert streamflow as much as 96.5 ft^3/s in summer months. Reach V–VI has no major surface-water diversions. It begins at the Verde Ditch return and continues to the Camp Verde gaging station (fig. 29).

Fish

Introduction and Background

The native fish of the Southwest are historically lacking in numbers and variety of species and are uniquely adapted to a range of conditions that include extreme temperatures, unpredictable flood flows, and sediment-laden waters. Historically, native fish frequently would experience periods with high flows or droughts that would locally diminish or extirpate populations, but following these major events there would be a relatively quick recolonization from tributary flow or isolated backwaters and pools. As a result, native fish continued to thrive into the 20th century. It was not until humans began to alter the landscape and introduce non-native species that native species began to decline and disappear. Specifically, native fish in the Verde River watershed have been declining since the introduction of non-native fish species nearly a century ago. Added to this are the reductions in base flow from increased water usage and the ongoing drought that began in the first decade of the 2000s. Together, these factors have led to rapid degradation of the in-stream habitat of native fish in the Verde River.

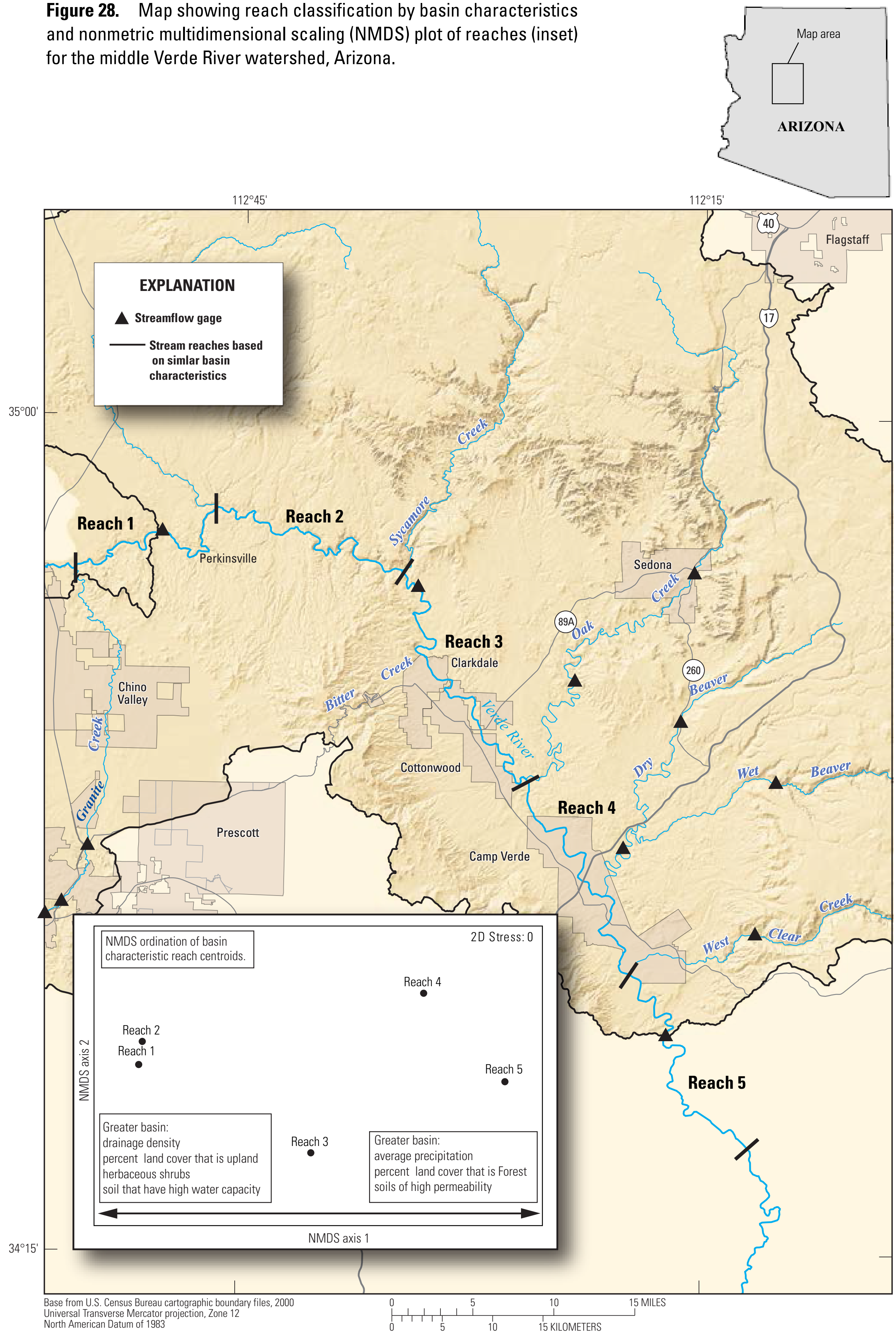
Adaptation of Native Fish to the Flow Regimes of Desert Streams

The flow regime in perennial streams in the American Southwest varies greatly seasonally and has shaped the evolution of native fish species (Deacon and Minckley, 1974). These perennial Southwestern streams have long periods near base flow, with higher flows and flooding associated with large-scale winter storms or more local summer monsoon storms that can cause flashy, intense floods. Though the months of snowmelt and the monsoon are predictable, the magnitude, timing, and rate of change are unpredictable (Bain and others, 1988; Poff and others, 1997). As a result, native fishes have become highly adapted to a varied flow regime (Lytle and Poff, 2004).

These fish can exhibit behavioral, morphological, or life-history traits that aid in their survival during flood and drought events (Lytle and Poff, 2004). When streams are regulated, peak flows decrease in magnitude and duration, and native fishes adapted to the natural flow regime may begin to decline in numbers. Non-native fishes, which are often adapted to moderate flows, are able to thrive in these circumstances, allowing them to prey on and outcompete native fishes (Bain and others, 1988; Poff and others, 1997).

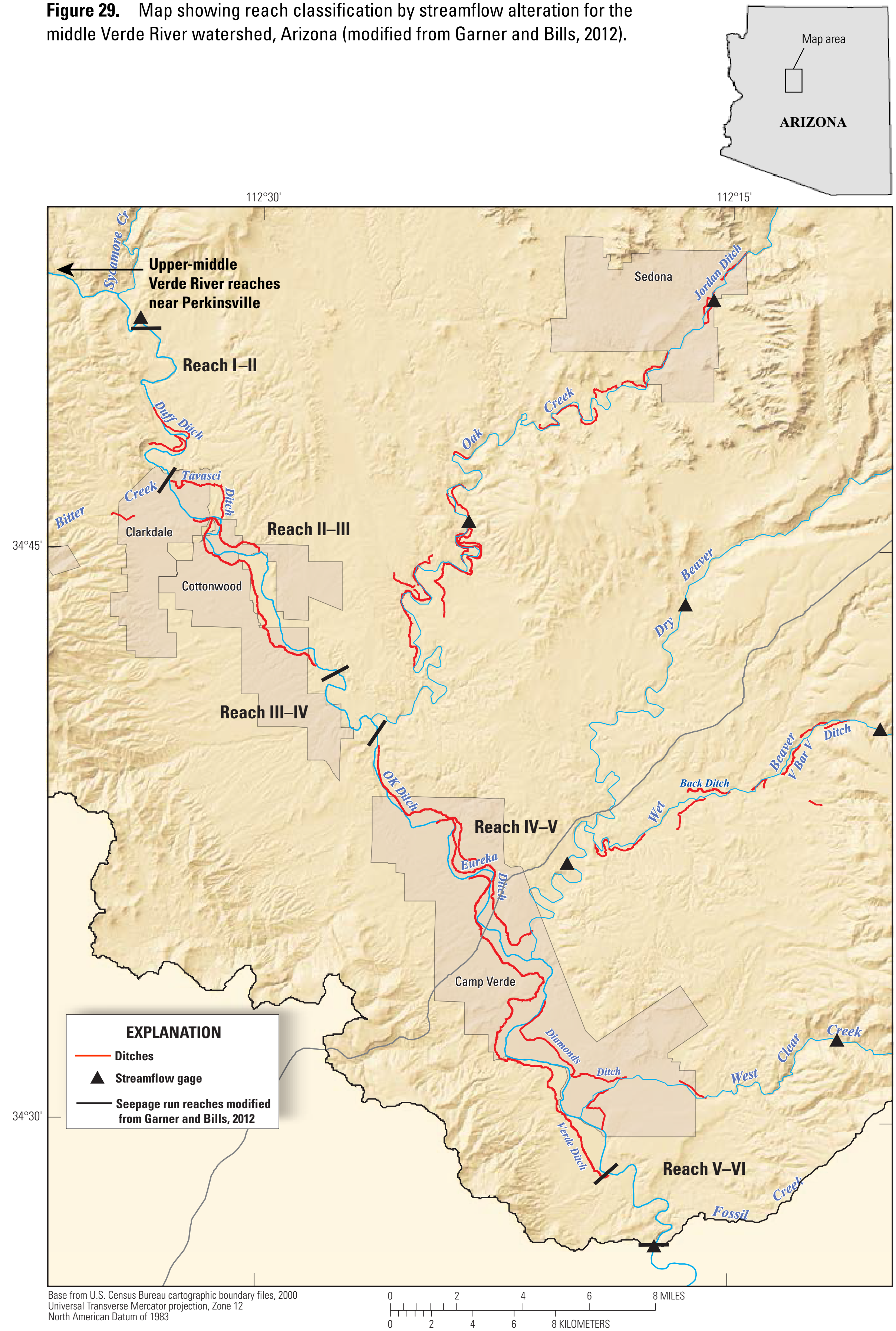
Physical factors are likely more important than biotic interactions in determining fish assemblages in a variable stream environment such as that in the semiarid Southwest.

Figure 28. Map showing reach classification by basin characteristics and nonmetric multidimensional scaling (NMDS) plot of reaches (inset) for the middle Verde River watershed, Arizona.



Base from U.S. Census Bureau cartographic boundary files, 2000
Universal Transverse Mercator projection, Zone 12
North American Datum of 1983

Figure 29. Map showing reach classification by streamflow alteration for the middle Verde River watershed, Arizona (modified from Garner and Bills, 2012).



However, biotic interactions become more important when streams become stable, such as in a regulated flow environment (Ross and others, 1985; Power and others, 1988). Stream-channel drying can exacerbate predation and competition as habitat area and complexity decline. When stream drying occurs, both abiotic and biotic factors change, which can in turn affect the species diversity and the species density. Many streams in the desert Southwest naturally go dry or flow is reduced to pools sometimes connected by shallow riffles or runs (Capone and Kushlan, 1991; Magoulick, 2000). Bigger fish tend to be found in the deeper waters of pools to avoid terrestrial, diving predators (Power, 1988), whereas small fish tend to stay in shallow water with complex habitat in which they hide to avoid swimming predators (Power, 1984; Harvey and Stewart, 1991; Magoulick, 2000). Harvey and Stewart (1991) have shown that as water volume and depth decline in drying streams, susceptibility of prey to predation increases. Zaret and Rand (1971) found that fish showed increased competition for food in drying stream pools in Panama due to overlap in habitat.

Increases in native fish populations have been documented after large floods in desert streams (Rinne, 2005; Rinne and Miller, 2006). One possible explanation is that these fish change their habitat use in the presence of higher velocity flow. Rinne (1989) found that the habitat use of the loach minnow (*Tiaroga cobitis*) in the Gila River was directly correlated with velocity. In higher velocities, the loach minnow tended to use larger substrate. Native desert fishes will essentially hide behind large substrate to avoid being swept downstream with the current. Other behavioral adaptations to flow include fish moving to safe stream margins when they sense increasing stream velocity (Bain and others, 1988; Lytle and Poff, 2004). In flume experiments comparing two fish species common in Arizona streams, the non-native western mosquito fish (*Gambusia affinis*) did not share the ability of the Sonoran topminnow (*Poeciliopsis occidentalis*) to react quickly to an increase in velocity and retreat to safer habitat (Meffe, 1984). In the short term, the ability to find low-velocity habitat in the stream during high flows or the ability to change substrate use during high flow could explain why native fishes in the Verde River increase in abundance following floods. Ultimately, as floods wash out non-native fish, this results in an increased reproductive success for native fish.

Non-native Fish of the Verde

Since the beginning of the 20th century, streams of the southwestern United States have been the focus of many efforts to introduce new fish species, and non-native fish continue to be stocked for sport fishing, as bait fish, and for biological control (Rinne and Janisch, 1995; Schade and Bonar, 2005; Lomnicky and others, 2007; Whittier and Peck, 2008). Intentional sport-fish stocking began toward the end of the 19th century with the introduction of common carp (*Cyprinis carpio*), channel catfish (*Ictalurus punctatus*), and various bass species (*Micropterus* spp.). Many more species have been introduced for sport fishing since then, including other catfish (*Ictalurus* spp., *Pylodictis* spp., *Ameiurus* spp.), northern pike (*Esox lucius*), crappies (*Pomoxis* spp.), sunfish (*Lepomis* spp.), rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*), and brook trout (*Salvelinus fontinalis*) (Rinne and Janisch, 1995). Those brought in as bait fish often include smaller species such as fathead minnow (*Pimephales promelas*), red shiner (*Cyprinella lutrensis*), and golden shiner (*Notemigonus crysoleucas*). The most notable case of a species brought in for biological control was the introduction of western mosquito fish (*Gambusia affinis*), which was introduced for the purpose of controlling mosquitos (Hubbs and others, 1991).

Non-native species are a threat to native species in the Verde River, as well as in other streams in the American Southwest (Minckley and Meffe, 1987; Rinne and Minckley, 1991), and have been since their introduction more than a century ago. Non-native fish have directly contributed to the decline and extirpation of native fish populations through predation, competition, and hybridization (Minckley and Marsh, 2009; Deacon, 1988; Miller and others 1989; Rinne and Miller, 2006; Schade and Bonar, 2005). For example, the western mosquitofish has eliminated the Sonoran topminnow in New Mexico and in many areas of Arizona (Hubbs and others, 1991) by preying on its eggs, larvae, and juveniles (Courtenay and Meffe, 1989). Both predation risk and competition can alter the behavior of native species by forcing the native species to move to less desirable habitat.

Rinne and Miller (2006) found a marked decrease in native fish accompanied by a significant increase in non-native fish in the upper-middle reach of the Verde River. They related this trend to changes in channel morphology, lower base flows, and reduced number of floods. A natural flow regime consisting of variable flows throughout the year can assist native populations by maintaining lower non-native populations (Rinne, 2005; Olden, 2006; Poff and others, 2007; Gido and Propst, 2012).

Native Fish of the Verde and Habitat Requirements

Fish taxonomic diversity has always been lower in the Southwest than in other parts of North America due to the unpredictable flow regimes found there (Moyle and Light, 1996; Minckley and Marsh, 2009). Rapid extirpation and recolonization is an adaptation of native desert fish, and interconnected fish populations within a larger basin are called metapopulations and are very important for species that may be locally extirpated. Metapopulations provide a source for recolonization following localized extirpations resulting from extreme flooding or drought conditions. Rinne and Miller (2006) state that the native fishes in southwestern desert streams consist of 95 percent Cypriniformes and 5 percent other taxa. Included in the Cypriniformes are the families Catostomidae (suckers) and Cyprinidae (minnows and dace).

Rinne (2005) and Rinne and Miller (2006) provide the most complete summary of native and non-native fish in the Verde River. A number of species had already been extirpated by the time comprehensive surveys were beginning in 1994. From 1994 to 2005, Rinne and Miller (2006) collected fish at seven sites in the upper-middle Verde River by electrofishing and seining and found the following six native species—desert sucker (*Catostomus clarki*), Sonora sucker (*Catostomus insignis*), longfin dace (*Agosia chrysogaster*), speckled dace (*Rhynichthys osculus*), roundtail chub (*Gila robusta*), and spikedace (*Meda fulgida*). Abundance estimates declined sharply during their population surveys. For example, longfin dace numbered more than 1,300 in 1994 and then were reduced by nearly three orders of magnitude by 1999. Desert sucker saw a similar decline, from 2,644 in 1994 to 126 by 1998. By 1997 non-native fish had become the dominant component in the Verde. A flood event occurred in 1995, but through the end of the Rinne and Miller (2006) study (through 2005), no further flooding occurred.

The life history of native fish in the Verde River watershed is closely related to the unpredictable hydrologic regime which is the primary determinant for ecosystem structure and function in desert aquatic ecosystems. Aquatic-habitat requirements and use will vary greatly by species and are related to morphological characteristics such as body size and shape, feeding habits, and spawning requirements (Lytle and Poff, 2004). Understanding the habitat characteristics required by native species is necessary for understanding how these species will be affected by habitat alteration related to changes in streamflow. In the following discussion regarding habitat use, we focus on the five common native species remaining in the Verde River—desert sucker, Sonora sucker, roundtail chub, speckled dace, and longfin dace (Rinne, 2005; Rinne and Miller, 2006).

Desert sucker young are generally observed in slow velocity gravel riffles, whereas the adults tend to feed and live in fast velocity riffles at night and use other types of riffle and pool habitat during the day (Rinne and Minckley, 1991; Rinne, 1992). The Sonora sucker spawns from late winter to early summer and requires coarse gravel for egg burial and hatching (Lee and others, 1981). These fish are found under banks, boulders, and debris in moderately shallow, sandy, gravelly pools during the day, and like the desert suckers, move to riffles and runs at night (Minckley, 1973; Lee and others, 1981; Rinne, 1992). The roundtail chub spawns in late spring to early summer. Chub typically occur in deep, slow water over a range of substrate associated with cover (Rees and others, 2005). The two dace species, longfin and speckled, are generally found in shallow water. Speckled dace typically occur in turbid high-velocity gravel/pebble riffles, whereas longfin dace use slower velocity riffles with sand/gravel substrates (Rinne, 1992).

There are three non-native fish species that have significant populations and impacts on native fish populations in the Verde River (Rinne, 2005)—red shiner, green sunfish (*Lepomis cyanellus*), and smallmouth bass (*Micropterus dolomieu*). Red shiner have been found to consume larvae of Cypriniformes (specifically roundtail chub, speckled dace, and bluehead sucker) in the Yampa and Green Rivers (Ruppert and others, 1993). None of the other non-native fishes examined, including sand shiner, channel catfish, and fathead minnow, consumed fish larvae. Red shiner occupy shallow areas used as nursery habitat by other fishes. Red shiner prefer invertebrates as prey but are opportunistic feeders and have been found to eat other fish (piscivorous) when preferred food was not readily available (Ruppert and others, 1993). Green sunfish are known to be piscivorous and do prey on chub species. Predation on Gila chub by green sunfish has been observed by Dudley and Matter (2000). In areas with green sunfish, mostly large adult Gila chub were observed, likely because they were too large for green sunfish to eat. Roundtail chub of all life stages have been found in Turkey Creek in New Mexico above falls that prevented the invasion of smallmouth bass, but below the falls only adult roundtail chub were found, suggesting juveniles were preyed on by smallmouth bass (Bestgen and Propst, 1989).

Magoulick (2000) studied the influence of abiotic factors on fish assemblages in the pools of drying streams and found that fish species and the size of the individuals observed in the pools were related to water depth and surface area, habitat heterogeneity, and density of canopy cover. Capone and Kushlan (1991) found that pool persistence in drying streams was an important factor in determining fish assemblage when streamflow returned. Abiotic factors have been shown to be more important in unstable environments than in homeostatic environments, where biotic pressures may be more influential on fish community structure (Matthews and Styron, 1981; Schlosser, 1982). Magoulick (2000) also indicated that pool depth is an important abiotic factor for maintaining native fish in streams that lose significant base flow and dewater significant parts of the stream reach. Small fish such as speckled dace and longfin dace immediately suffer from the loss of shallow riffle habitat, being forced into pools with larger predator species (Haney and others, 2008). If the pools are deep and contain cover, then the risk of predation will be less, whereas if the pools are shallow, predation from large predators is likely imminent. Similarly, larger fish would suffer from a lack of deep pools by being unable to hide from terrestrial predators. Should base flow continue to decrease in the Verde River, several of native desert fishes could be at an even greater risk for extirpation. Fagan and others (2002) found that habitat fragmentation of Sonoran Desert fishes was much more likely to increase extinction risk than population declines alone. In fact, the number of species' occurrences had no additional effect on extinction risk once fragmentation had been factored out.

Fish assemblages are also affected by high-flow events. Rinne (2005) demonstrated that native fish species showed significant population growth during high flow years, and this has also been documented elsewhere in the Southwest. In the San Juan River, Gido and Propst (2004) found that native dace and sucker species increased in numbers in years following high spring discharge (greater than 494 ft^3/s) and non-native species, such as red shiner, Western mosquito fish, and common carp, benefited from low flow periods and declined in numbers during the high flows. Non-native species are not adapted to the flash floods that commonly occur during the North American summer monsoon. High flows are one of the last remaining natural defenses native fish have to non-native fish pressures.

Temporal and Spatial Considerations of Fish Data Collected in the Middle Verde Watershed

After reviewing fish-population surveys conducted by different agencies and evaluating the differences in sampling efforts among agencies, sampling efforts were determined to be mostly similar with a majority of the data provided by AZGFD. Independent of the timing of collection, the assemblage data showed interesting spatial patterns with greater percentages of native-fish populations in the upper-middle river reaches (fig. 25) than in the lower-middle reaches. However, there were areas upstream of Perkinsville that had lower percentages of native fish, similar to parts of the river adjacent to Cottonwood and Camp Verde. To determine the cause of this spatial pattern, other factors and sampling conditions must be considered (for example, timing, location, habitat, and flow conditions; fig. 30). The data collection (population surveys) distributions (fig. 30) indicate that more than half of the fish-population surveys occurred between 2004 and 2008 and during the summer months (June–August) with moderate to no flooding conditions before sampling. More than half the collections occurred in the upper-middle Verde River (drainage area 2,000–3,000 square miles) in run habitat that was minimally affected by diversions.

Some fish collections did occur following large flow events and during extended periods of low flow (fig. 27). The duration of beneficial effects to native fish following a large flood are largely unknown, as is the magnitude that is necessary to initiate these beneficial conditions. Following a large flood in 1995 that exceeded 20,000 ft^3/s (peak flow at Clarkdale gage), median percentages of native fish made up almost 50 percent of the total fish population but then declined by about two-thirds by 2002 until 2005 when another large flow occurred exceeding 25,000 ft^3/s (peak flow at Clarkdale gage; fig. 31). Following the 2005 event, the native-fish percentages rebounded until multiple years of moderate to low magnitude flooding conditions reduced the median native-fish percentages to less than 10 percent in 2009. The years of moderate flooding had variable effects on the percentages of native fish.

The timing, flow regime, and location of the fish surveys need to be considered when trying to understand the effects of flow alteration, although data can be difficult to interpret because of the large number of confounding human and environmental factors. One way to reduce the variability caused by environmental factors is to classify stream reaches based on basin characteristics. This classification scheme will be used first (fig. 28) and then followed by an evaluation using the flow-alteration reach classification scheme (fig. 29).

Distribution of Fish Using Basin Characteristic Reach Characterization

General spatial patterns, independent of time, from the 312 fish surveys conducted between 1992 and 2011 indicate that native species generally were a greater percentage of the total fish population in the upper-middle than in the lower-middle Verde River and decreased with downstream distance (fig. 25). Similar patterns were reported by Rinne (2005), and this pattern may be related to many factors including flow alteration (surface-water diversions and ground-water pumping), habitat, water quality, water chemistry, and (or) flow variability and volume (Rinne and Miller, 2006). It is likely that declining native-fish populations are related to a combination of stressors, including both anthropogenic factors and pressure from non-native fish through competition and predation. There were negative correlations between native-sucker families and non-native sunfish, minnow, and catfish families ($\tau = -0.53$, -0.14, -0.17; all $p < 0.001$). The NMDS ordination of species data showed that certain non-native species occurred in similar abundances and in similar locations with native species. Native and non-native fish regularly interact, suggesting that they compete for the same habitat resources (fig. 32). Predatory catfish and sunfish clustered near the native suckers and chub. Such use of similar habitat increases competition and the potential for predation. Similarly, the native-dace species cluster near red shiners and mosquito fish.

The five reaches (defined by basin characteristics) of the upper-middle and lower-middle Verde River described previously were used to compare the composition of fish assemblages (fig. 33). The reach designation showed that native fish were greater in the upper reaches of the watershed, where reaches are characterized with smaller drainages, less precipitation, and less flow. The trend of native-fish abundance decreasing with downstream distance was apparent as well. The median abundance of native fish decreased about 50 percent from reach 1 to reach 5 (fig. 33*A*). The median percentage of native fish decreased by about 10–15 percent from reach 1 to 2, 15–20 percent from reach 2 to 3, and 20–25 percent from reach 4 to 5. Differences were only statistically significant from reach 1 to 3 and reach 4 to 5. The percentages of native fish, as well as family composition (fig. 33*B*), were most similar in reaches 3 and 4, although the NMDS indicated larger differences in the basin characteristics than the other adjacent reaches, 1–2 and 4–5. The similarities in fish composition between reaches 3 and 4 may be related to the effects of basin urbanization and diversions, and these effects potentially predominate in these reaches. Land-use changes can alter the stream habitat by reducing flow and increasing sedimentation, and these habitat changes can lead to an increase in non-native fish species, such as bass and sunfish, that tend to spawn in low-velocity, pool habitats (Coles and others, 2012).

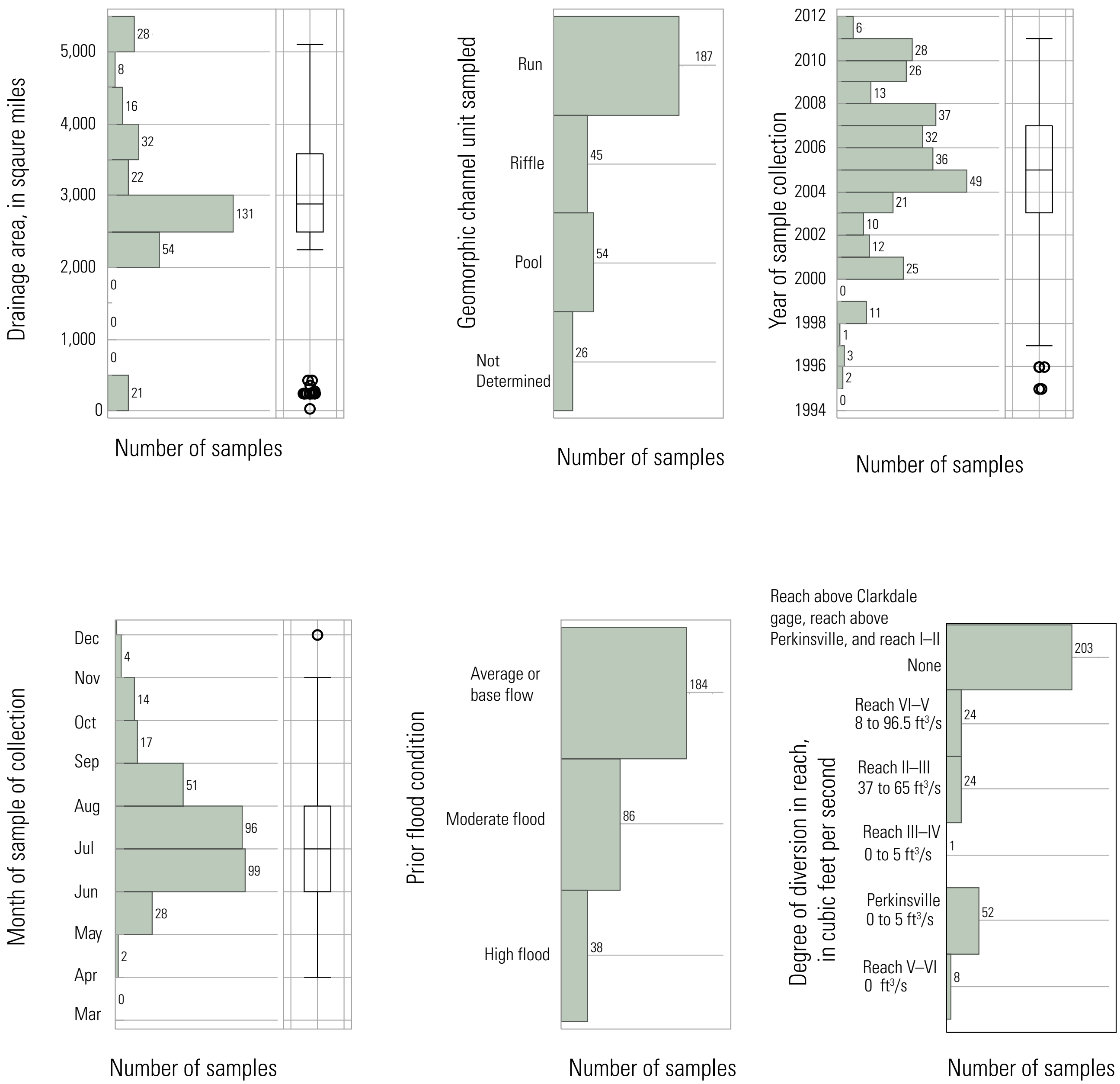


Figure 30. Plots showing the distribution of fish-survey data collection for the middle Verde River watershed, Arizona, by drainage area, geomorphic channel units, year of collection, month of collection, flow condition, and degree of diversion. ft^3/s, cubic feet per second.

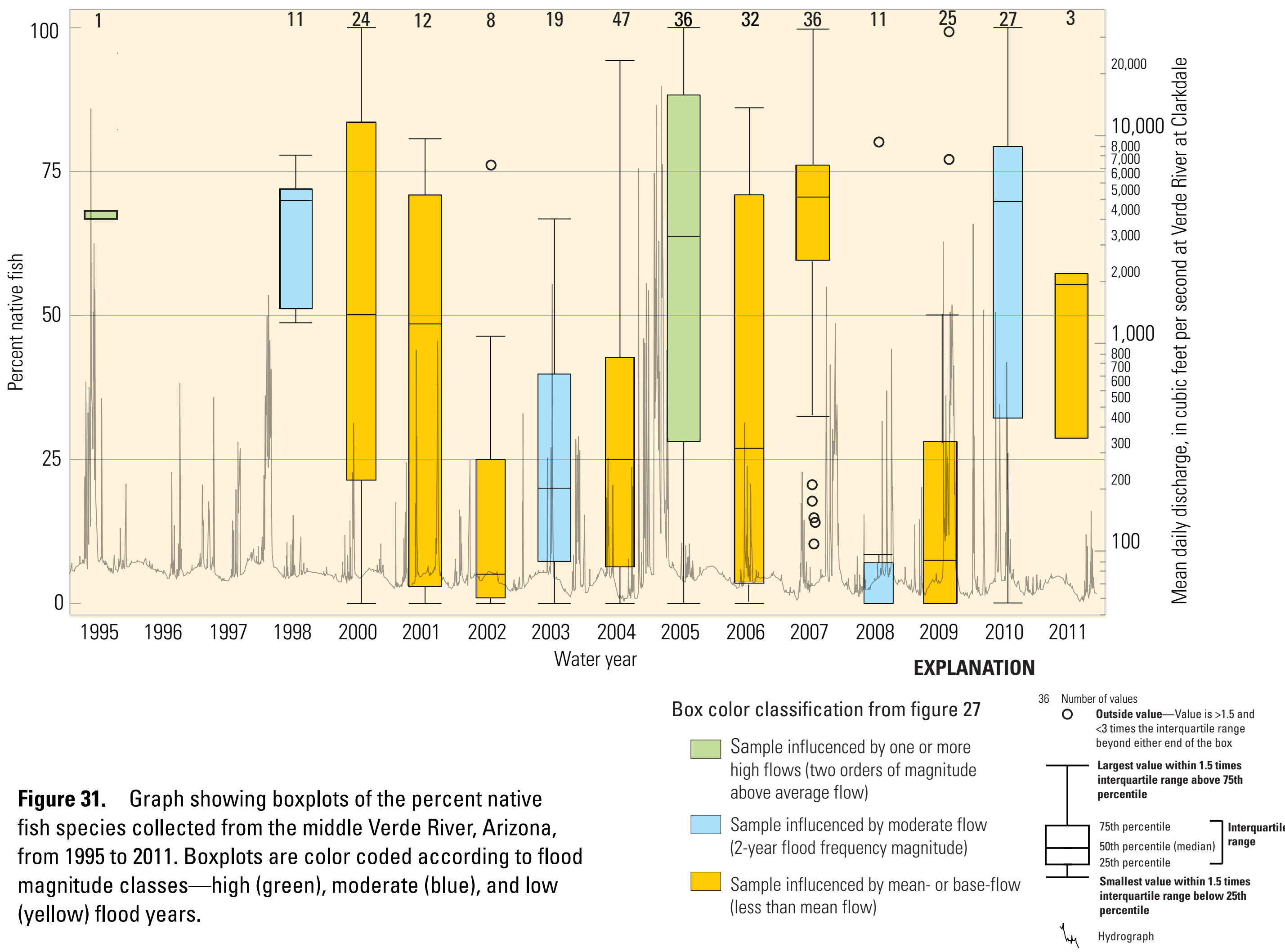


Figure 31. Graph showing boxplots of the percent native fish species collected from the middle Verde River, Arizona, from 1995 to 2011. Boxplots are color coded according to flood magnitude classes—high (green), moderate (blue), and low (yellow) flood years.

The transition to greater proportions of sunfish started in reach 2, where the percentage is similar to suckers and becomes significantly greater by reach 3. At reach 5 the median percent of sunfish is 10 times that of the suckers (fig. 33*B*). The minnow and catfish families stayed relatively consistent throughout the entire middle Verde River reach (from 2 to 10 percent), and the mosquito fish and rainbow trout (not shown) had very minor contributions. The feeding guilds in reach 1 and 2 are more evenly distributed than the other reaches. Piscivores were about twice as common as other feeding groups in reaches 3, 4, and 5 where the piscivore feeding guild was made up primarily of sunfish species (fig. 33*C*). Predation from non-native fish is among the many stressors affecting native fish populations in the Verde River.

The patterns in the NMDS ordination supported that specific Verde River reaches were an important factor in fish assemblages (fig. 34). Fish assemblages for reaches 1 and 2 (or the upper-middle Verde River) are grouped separately from fish in reaches 3, 4, and 5. The native sucker families and the catfish were more strongly associated with reaches 1 and 2, whereas the sunfish were more closely associated with reaches, 3, 4, and 5. Reach 5 had the lowest proportion of native fish, and fish in this reach plotted furthest away from all other reaches.

Geomorphic channel units were assigned to each Verde River sampling location using aerial imagery. One of three class options, pool, run, or riffle, was assigned to each sampling location on the Verde River. This type of habitat characterization is a general estimation of geomorphic descriptors, although results appeared to be sensitive enough to distinguish between flow-substrate and presence of different families (fig. 35*A*). The median of percent native fish was significantly greater (p=0.0070, Wilcoxon rank sums pairwise test) in stream reaches classified as run than in those classified as pool. Non-native fish were more often found using pool habitat (p=0.0022, Wilcoxon rank sums pairwise test) than native fish. Family differences reflected the differences among native and non-native fish, because all suckers are native (found most often in runs) and all the sunfish are non-native (most often found in pools, fig. 35*B*). The minnow and catfish families were found in all pool, riffle, and run habitat equally.

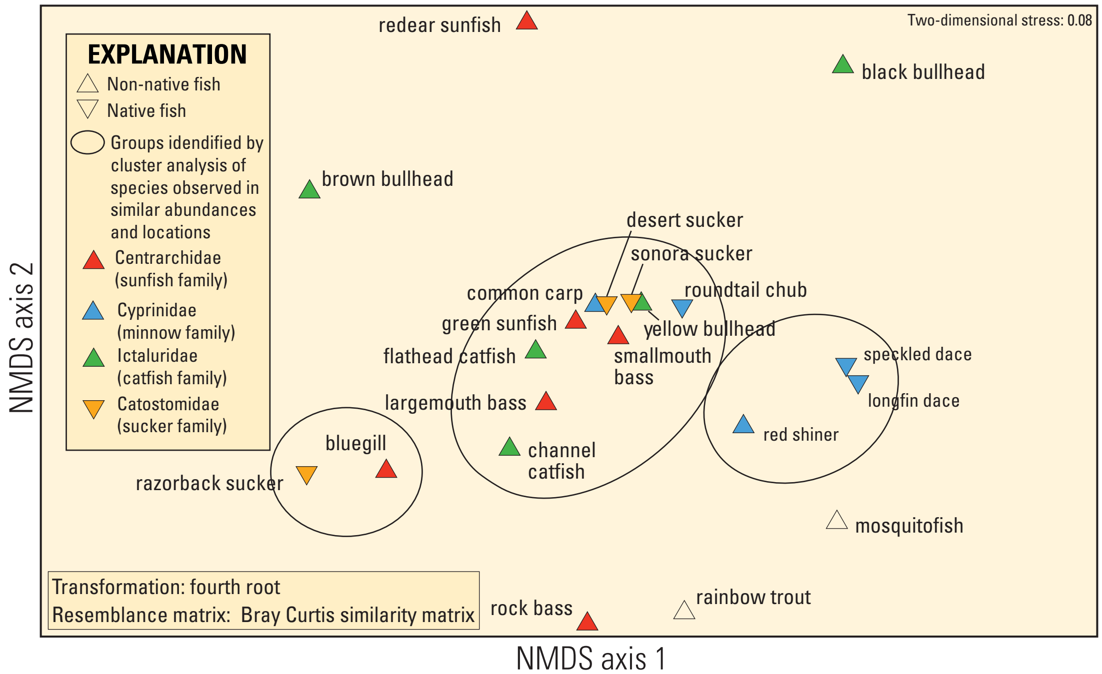


Figure 32. Diagram showing nonmetric multidimensional scaling (NMDS) ordination of fish-species similarities across all samples from the middle Verde River, Arizona. Species are identified by family (except for trout and mosquitofish) and native and non-native identification. Significant clusters are highlighted with circle outlines. Common and scientific names for native fish species: desert sucker (*Catostomus clarki*), razorback sucker (*Xyrauchen texanus*), Sonora sucker (*Catostomus insignis*), roundtail chub (*Gila robusta*), speckled dace (*Rhinichthys osculus*), longfin dace (*Agosia chrysogaster*). Common and scientific names for non-native fish species: largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieu*), rock bass (*Ambloplites rupestris*), green sunfish (*Lepomis cyanellus*), redear sunfish (*Lepomis microlophus*), brown bullhead (*Ameiurus nebulosus*), black bullhead (*Ameiurus melas*), yellow bullhead (*Ameiurus natalis*), channel catfish (*Ictalurus punctatus*), flathead catfish (*Pylodictis olivaris*), common carp (*Cyprinus carpio*), red shiner (*Cyprinella lutrensis*), rainbow trout (*Oncorhynchus mykiss*), western mosquitofish (*Gambusia affinis*).

Distribution of Fish Using Flow-Alteration Reach Characterization

In the second scheme, reaches were defined by degree of alteration from ditches and diversion (Garner and Bills, 2012). Fish data collected in the reaches of the lower-middle Verde River were also compared to fish data collected in the upper-middle Verde River watershed, above and below Perkinsville (minor diversions below Perkinsville, see fig. 28) and the major tributaries to the Verde River—Oak Creek, Wet Beaver Creek, and West Clear Creek (fig. 29). Fish data were collected over the course of many years and during different months.

There was a significant decrease in the median percentage of native fish from the upper-middle Verde River reaches, where fewer diversions exist, to reach II–III, which is characterized as a highly diverted reach (p<0.001, Wilcoxon pairwise test) (fig. 36*A*). Reach III–IV is a short section of river where diverted water is returned to the river and no other major diversions exist. In this nondiverted reach, the median percentage of native fish increased about 20 percent. In reach IV–V, where water is diverted to the OK, Eureka, Verde, and Diamond S ditches the median percentage of native fish then decreased by about 10 percent and was significantly less than in the upper-middle reaches. The percentage of native fish continued to decrease in reach V–VI (p<0.001, Wilcoxon pairwise test), at which point a large volume of water has been diverted from the Verde River. Native-fish percentage was significantly less here than in any of the other reaches further upstream. In the undiverted upper reaches of the Middle Verde River watershed, native suckers made up the greatest proportion of fish assemblages (fig. 36*B*). In the downstream reaches of the river, where there are more diversions, the proportion of suckers was reduced and the percentage of non-native sunfish increased. However, the median percentage of minnows and catfish did not increase beyond that of the suckers until reach V-VI.

The NMDS ordination of the reach-assemblage multivariate centroids (fig. 37*A*) supports the changes observed in fish-assemblage composition when proportions of native fish are evaluated from upstream to downstream along the middle Verde River. Note that figure 37*B* and figure 37*C* are the same NMDS ordination plot, but different categorical information is superimposed on to each assemblage. In fig. 37*B*,

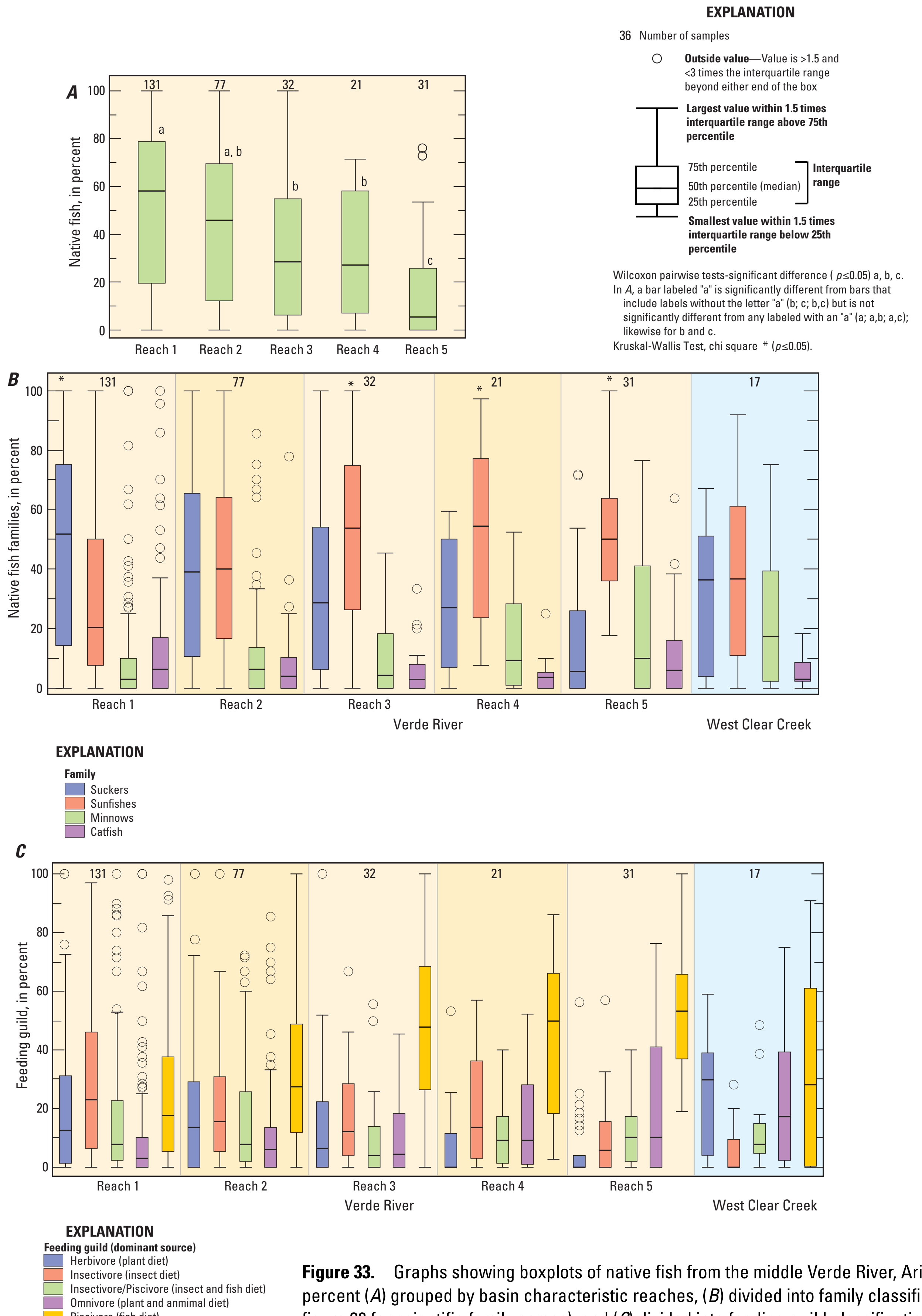


Figure 33. Graphs showing boxplots of native fish from the middle Verde River, Arizona, in percent (*A*) grouped by basin characteristic reaches, (*B*) divided into family classifications (see figure 32 for scientific family names), and (*C*) divided into feeding-guild classifications.

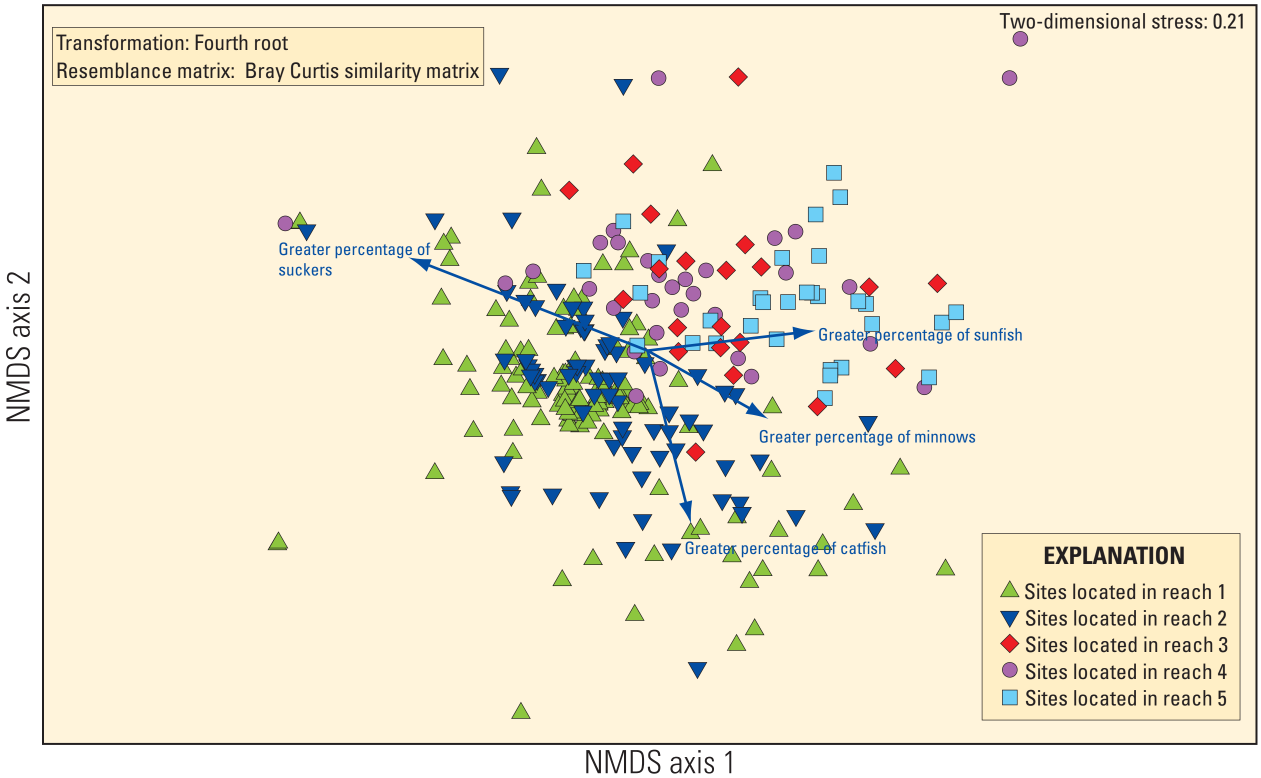


Figure 34. Diagram showing nonmetric multidimensional scaling (NMDS) ordination for fish assemblages labeled by basin characteristic reaches for the middle Verde River, Arizona. Fish families are shown as correlation vectors (see figure 32 for scientific family names).

the assemblage for each location in the seepage-run is color coded by reach and related to correlated species; in fig. 37*C*, the color coding was assigned by the degree of diversion and correlated with four dominant fish families. West Clear Creek, which joins the Verde River upstream of reach V–VI, was included to demonstrate where a mostly undiverted tributary (above the West Clear Creek campground, fig. 29) would plot in relation to the Verde River reaches (fig. 37*B*). Although West Clear Creek fish assemblages were more similar to the minimally diverted reaches of the upper-middle Verde River than to diverted reaches in the lower-middle Verde River, there are also habitat considerations that contribute to the similarities in assemblages (fig. 37*C*). The percentage of non-native sunfish was more correlated with reaches having significant surface-water diversions than were percentages of either native suckers or minnows. There was also overlap of non-native fish—mostly largemouth bass, channel catfish, flathead catfish, and green sunfish—in the diverted reaches II–III and V–VI.

The percent suckers and percent sunfish and overall abundance of each as reported in the available studies were plotted against the location (river mile) where they were observed along the middle Verde River. There was a statistically significant decrease in the percentage and abundance of suckers and a statistically significant increase in the percentage of sunfish with distance downstream (fig. 38*A*, *B*). In the upper-middle reaches, suckers were often at or above 50 percent of the fish population, whereas in the lower-middle reaches, suckers were mostly below 50 percent and sunfish were consistently at or above 50 percent of the population (fig. 38*C*). Samples collected after large and moderate floods in diverted reaches were limited, but there is some evidence to suggest that in highly diverted reaches during larger flood years, native sucker abundance was lower than in undiverted reaches (fig. 38*B*). Below the most highly diverted reaches (II–III and IV–V), the percentage of suckers was significantly lower than the percentage of sunfish.

Relation of Flow Metrics to Fish Community Metrics and Assemblages to Flow Metrics

Fish metrics were calculated for a subset of samples that were near a streamgage, and flow metrics were computed for 30, 90, and 365 days before the collection. Correlation scatter plots were used to identify significant relations using a Kendall's tau coefficient ($p \leq 0.05$). Overall, most flow and fish metric correlations showed weak associations ($\tau < 0.3$).

Fish total abundance (total number of all fish collected) was the primary fish metric associated with the 30-day flow metrics. More significant associations were identified with the 365-day flow metrics (fig. 39) than with the 30-day or 90-day flow metrics. There were also more negative correlations with the 365-day flow metrics (particularly with minnows and

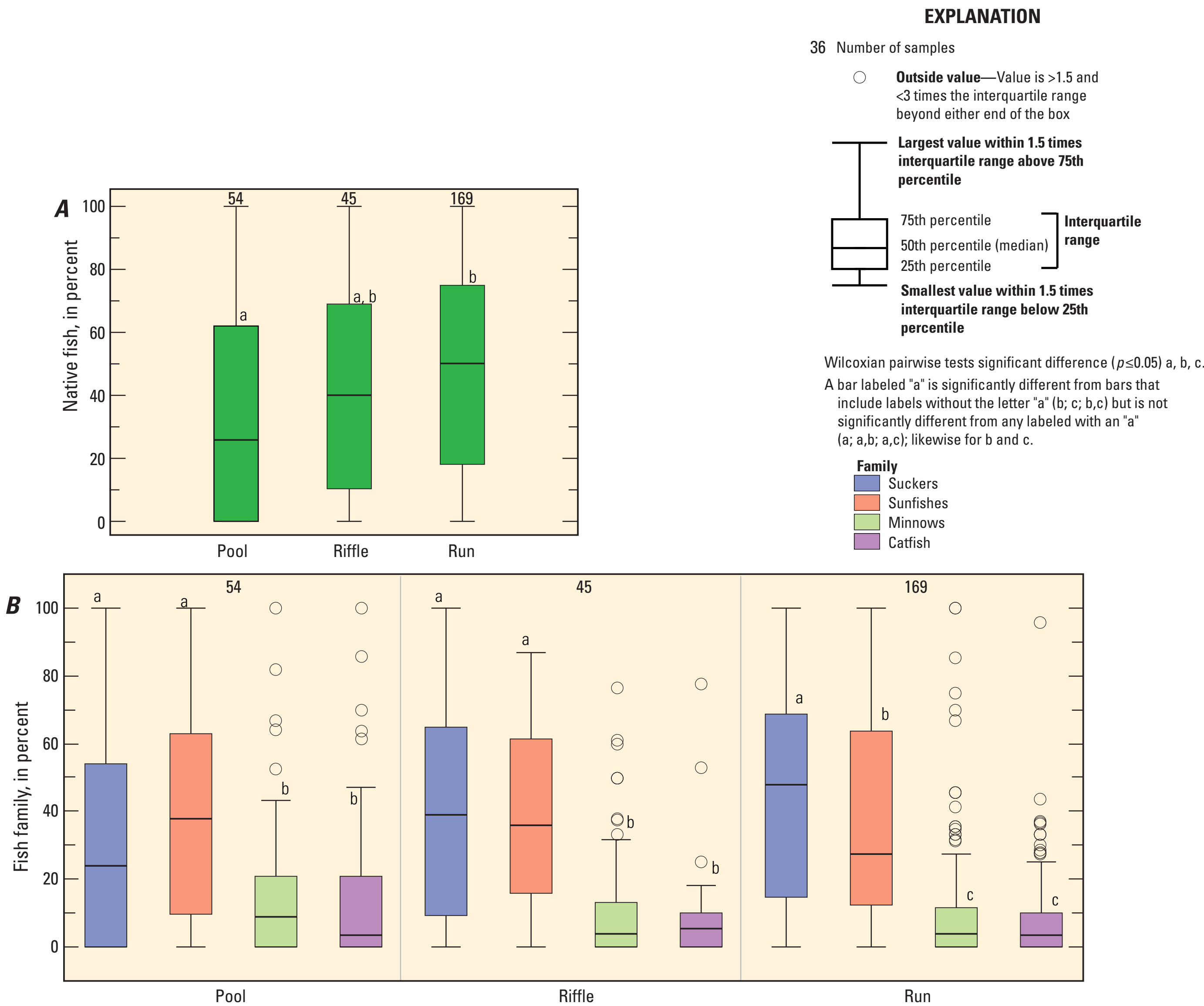


Figure 35. Graphs showing boxplots of (*A*) percent native fish for each geomorphic channel unit (GCU-pool, riffle, run) classification and (*B*) each family percentage (see figure 32 for scientific family names) within each GCU classification for samples collected on the middle Verde River, Arizona.

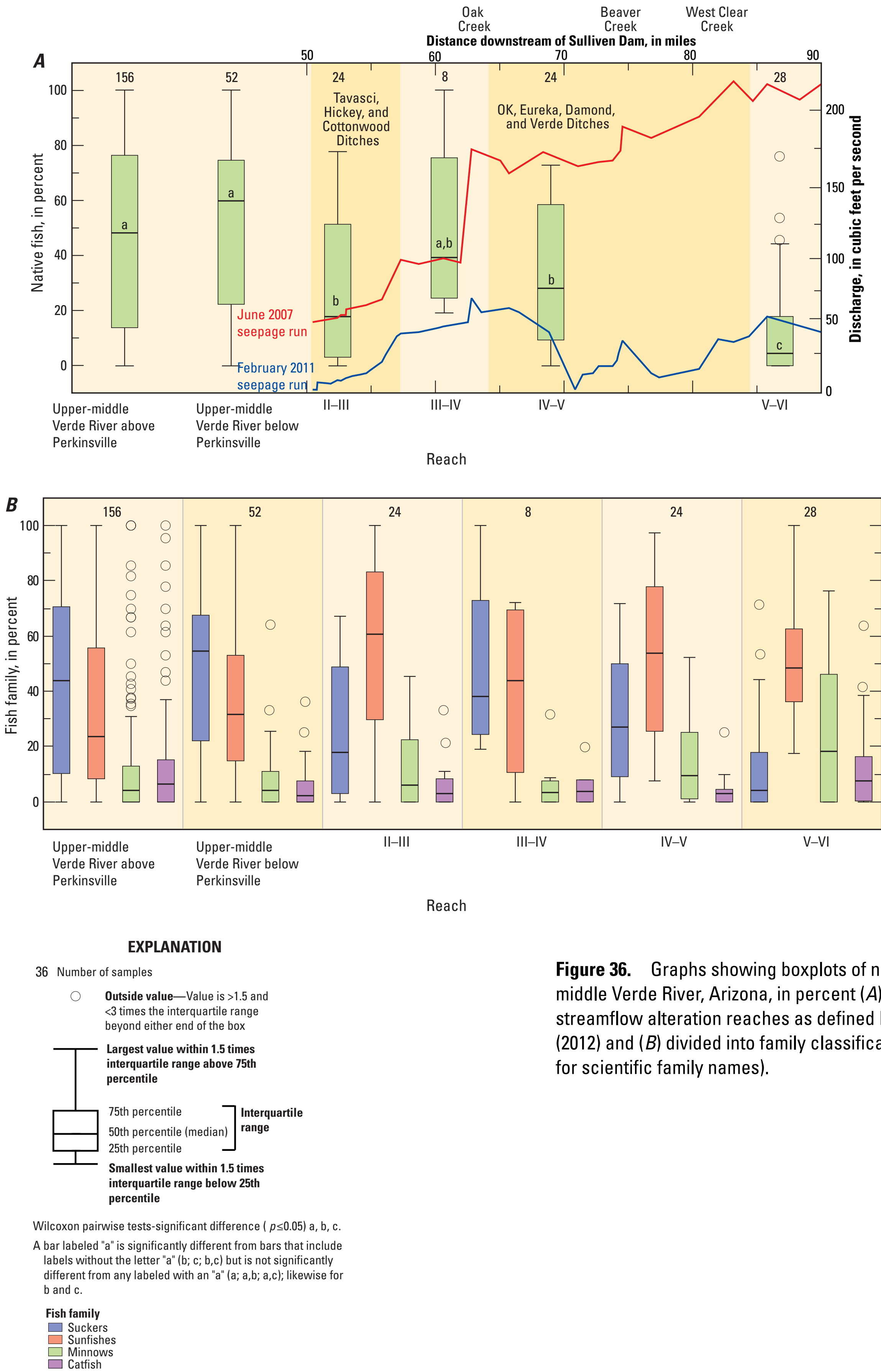


Figure 36. Graphs showing boxplots of native fish in the middle Verde River, Arizona, in percent (*A*) and grouped by streamflow alteration reaches as defined by Garner and Bills (2012) and (*B*) divided into family classifications (see figure 32 for scientific family names).

catfish) than with the 90-day metrics which were generally positively associated with fish metrics (particularly sunfish). More responsive (significant relation) high-flow metrics included those associated with magnitude and frequency, such as maximum monthly flow, median annual number of high-flow events, and median annual maximum streamflow. More responsive low-flow metrics included those associated with flow magnitude and frequency of discrete low-flow periods, including minimum monthly flow and median annual number of low-flow periods. Other responsive flow metrics were those describing the rate of change or the variable flow conditions related to flooding "flashiness" of the streamflow. These include the base-flow recession coefficient, 50th percentile of absolute value of percent daily change, and coefficient of variation of the log-transformed daily flows.

Several flow metrics were plotted against the fish metrics using quantile regression (figs. 40 and 41). Significant ceiling and floor thresholds (quantile thresholds at the largest quantile with narrowest confidence intervals for regression line slope) were analyzed for directionality of the association with the fish metrics. For the 365-day metrics, the percent of native suckers responded more positively to the ratio of the median annual minimum daily streamflow and median streamflow (Q_{min}/Q_{50}; fig. 40*A*) at the higher quantiles. At about the 0.80 quantile was the greatest slope and approximate threshold of the sample distribution. This metric is a measure of seasonal streamflow variation during periods of low flow. The sucker response was negative to the median annual number of low-flow periods (fig. 40*B*), but the slopes were an order of magnitude less than Q_{min}/Q_{50} slopes indicating about the same response (2–5 percent decrease) from 1 to 6.5 low-flow events. The greatest response (slopes) was similar between the 0.50, 0.75, and 0.90 quantiles. This suggests that suckers are sensitive to variability in low-flow periods and are somewhat less sensitive to the number of periods. Sunfish population proportions, which are negatively correlated to sucker proportions, responded positively to the minimum monthly streamflow (fig. 40*C*). This response was greatest at the 0.50-quantile threshold, and slopes were relatively small, about a 0.5–1 percent response change. Although sunfish respond more positively to higher minimum monthly streamflows, suckers (not shown in graph) appear to have a greater upward response at lower minimum values but then decrease as the minimum monthly streamflow increases.

The significant fish-metric responses to the 90-day flow metrics were primarily related to variability (fig. 41). The inverse relation between suckers and sunfish is more prominent in the 90-day flow metric relations (suckers, fig. 41*A* and 41*B*; sunfish, fig. 41*C* and *D*). For example, the percentage of suckers had negative responses with greater slopes for the 0.75–0.90 quantiles for the hydrologic metrics—coefficient of variation of log-transformed daily streamflow (fig. 41*A*) and the normalized range of monthly streamflow (fig. 41*B*). The normalized range of monthly flow (NRMF) is the difference between the maximum and minimum monthly flow each year divided by the mean flow for that year. The percentage of sunfish increased with increasing coefficient of variation of log-transformed daily flows and NRMF (fig. 41*C* and 41*D*, respectively), but the greatest slopes occurred at about the 0.60 coefficient of variation of log-transformed daily flows and 0.30 (NRMF) quantiles. These two metrics represent measures of daily and seasonal variability, respectively. With the relation of coefficient of variation of log-transformed daily flows, the percentages of suckers decrease by more than 50 percent for three quarters of the sample distribution (fig.41*A*), whereas the sunfish distribution increases by 10–20 percent as coefficient of variation of log-transformed daily flows increases. The coefficient of variation of log-transformed daily flows may be susceptible to differing results (slope direction change) depending on the time interval used for calculation of flow metrics. A similar pattern to coefficient of variation of log-transformed daily flows is occurring with suckers and sunfish in relation to the NRMF metrics suggesting that NRMF and coefficient of variation of log-transformed daily flows might be correlated because they are both measures of variability.

The relations among flow metrics, diversions, and other stressors, such as non-native fish is a complex and dynamic one. It can be difficult to detect the signal of flow alterations within the noise (variability) created by confounding variables in large scale studies. West Clear Creek is the only tributary with fish-assemblage data collected over several years and no flow alteration. The fish community has changed since the early 1990s to a more introduced-species dominated stream. A NMDS ordination shows how assemblages have changed over time, with the percentage of native fish declining in the early 2000s but increasing in samples collected in 2004, 2007, 2008, and 2011 (fig. 42*A*). Figure 42*B* shows the same NMDS ordination but with median number of high-flow events plotted instead of native fish. The high-flow overlay suggests the increased number of flows may be influencing the community shifts in the fish assemblages from 1995–2000 to 2009–2011.

Pilot Study—Native and Non-native Fish Microhabitat

A fish microhabitat pilot study was developed and executed to investigate native and non-native fish and their associations to microhabitat and flow characteristics in the Verde River and its major tributaries. The main objectives of the pilot study were to compare abundance and distribution of native and non-native fish at five sites in the Verde River watershed and to describe microhabitat use by native and non-native fish. With a protocol for understanding microhabitat requirements of aquatic biota developed for the five sites, it can then be used in the future to understand microhabitat requirements in other areas in the upper-middle and lower-middle Verde River.

The life histories of native fish in the Verde River watershed are closely related to the unpredictable hydrologic regime. This is the primary determinant for ecosystem structure and function in desert aquatic ecosystems. Aquatic microhabitat requirements and use will vary greatly by species, and

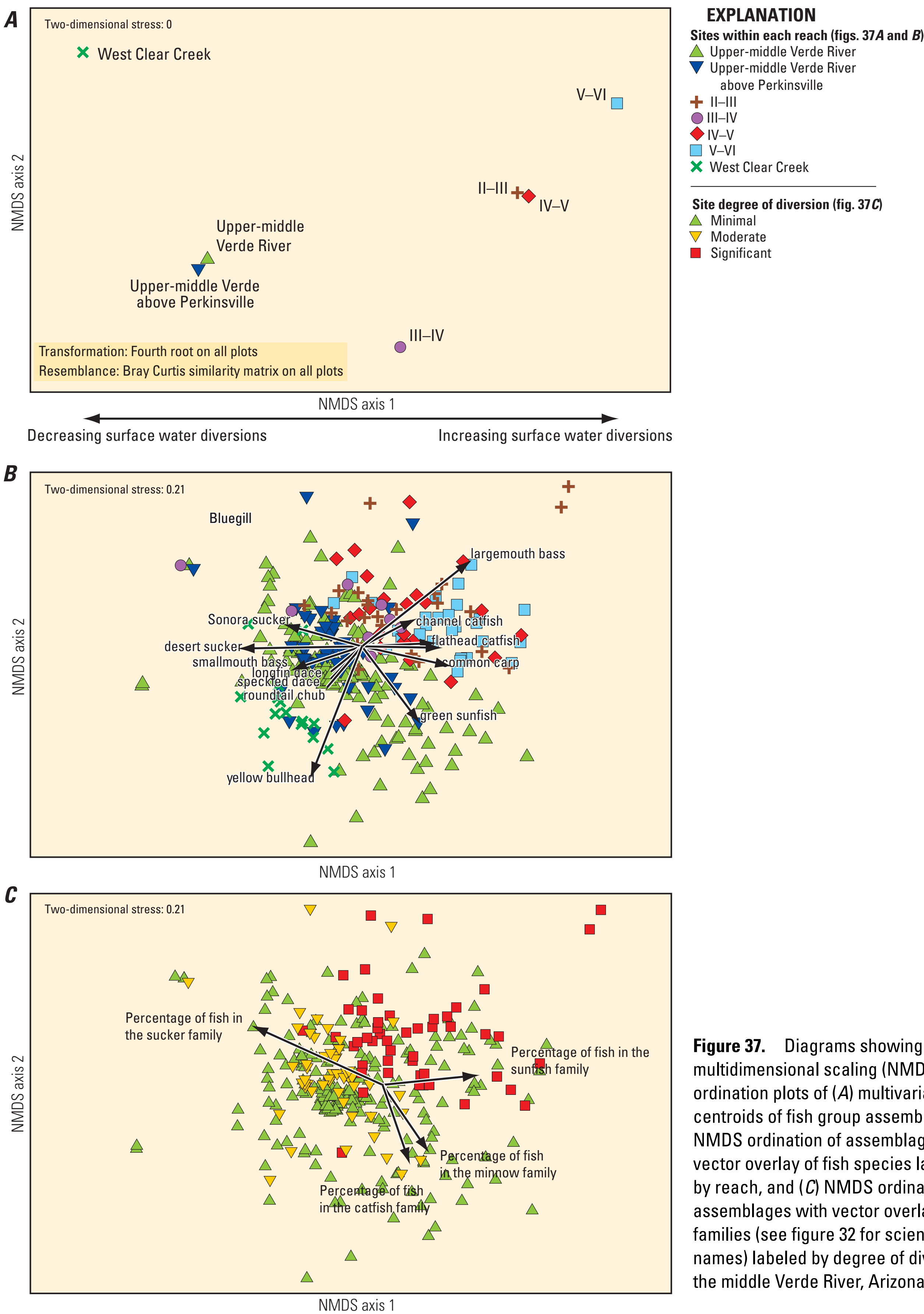


Figure 37. Diagrams showing nonmetric multidimensional scaling (NMDS) ordination plots of (*A*) multivariate centroids of fish group assemblages, (*B*) NMDS ordination of assemblages with vector overlay of fish species labeled by reach, and (*C*) NMDS ordination of assemblages with vector overlay of fish families (see figure 32 for scientific fish names) labeled by degree of diversion for the middle Verde River, Arizona.

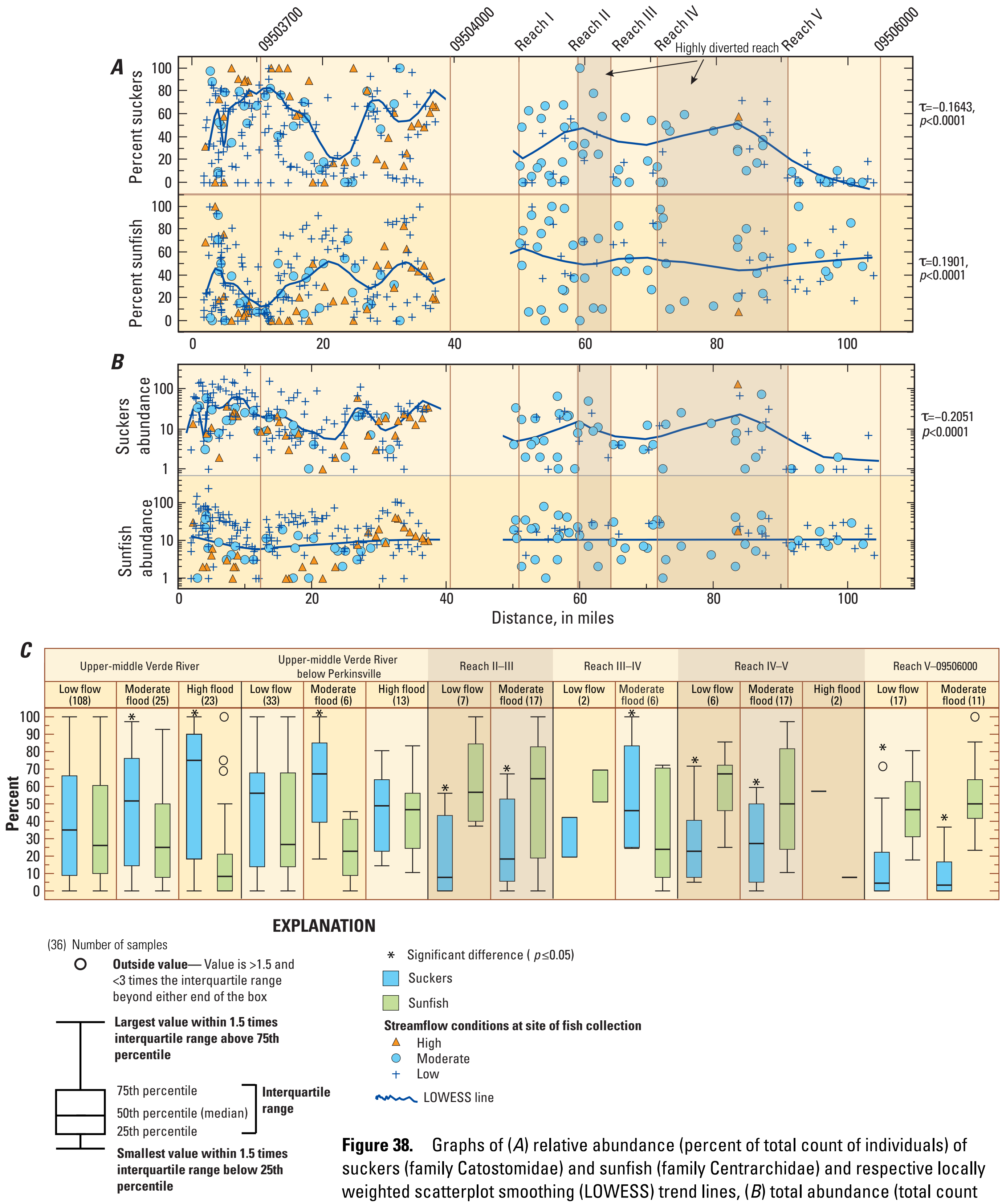


Figure 38. Graphs of (*A*) relative abundance (percent of total count of individuals) of suckers (family Catostomidae) and sunfish (family Centrarchidae) and respective locally weighted scatterplot smoothing (LOWESS) trend lines, (*B*) total abundance (total count of individuals in family) of suckers and sunfish and LOWESS trend lines, and (*C*) relative abundance of suckers and sunfish plotted as boxplots categorized by reach and flood condition for the middle Verde River, Arizona. 09503700, Verde River near Paulden; 09504000, Verde River near Clarkdale, AZ; 09506000, Verde River near Camp Verde, AZ. τ, Kendall's tau correlation coefficient presented for trend significance at $p<0.05$.

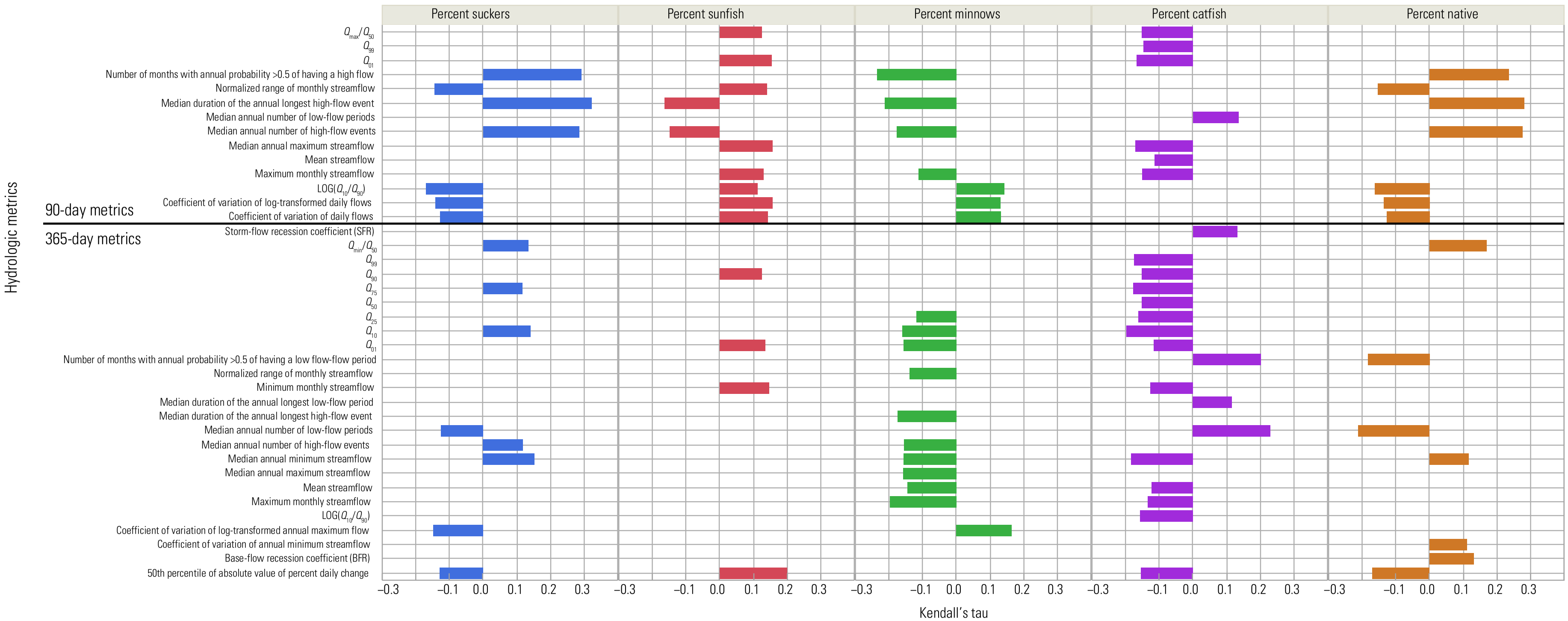
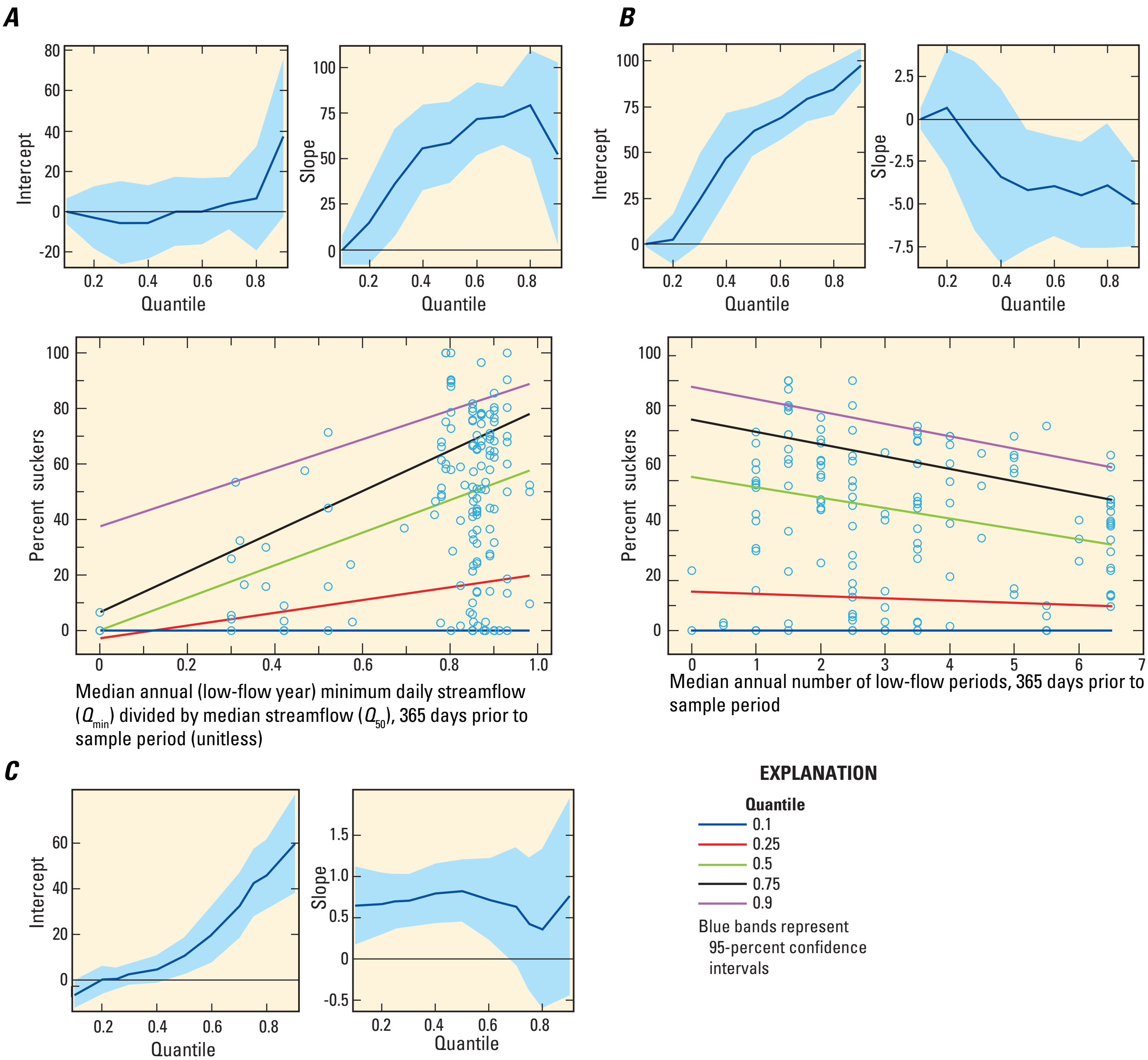


Figure 39. Graph of significant correlations (Kendall's tau (τ) correlation coefficient, $p \leq 0.05$) between 90- and 365-day hydrologic metrics and select fish metrics for the for the middle Verde River, Arizona. See figure 32 for scientific family names.



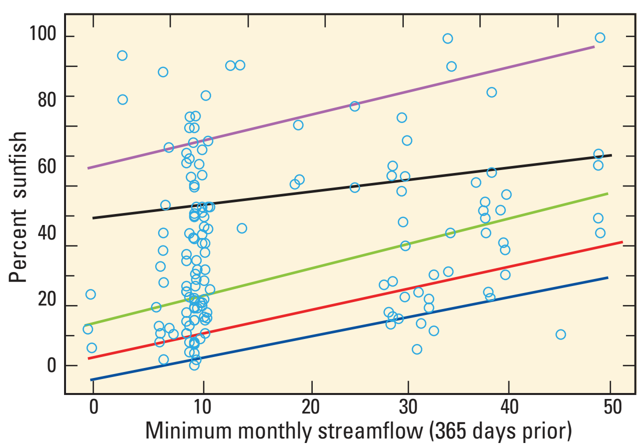
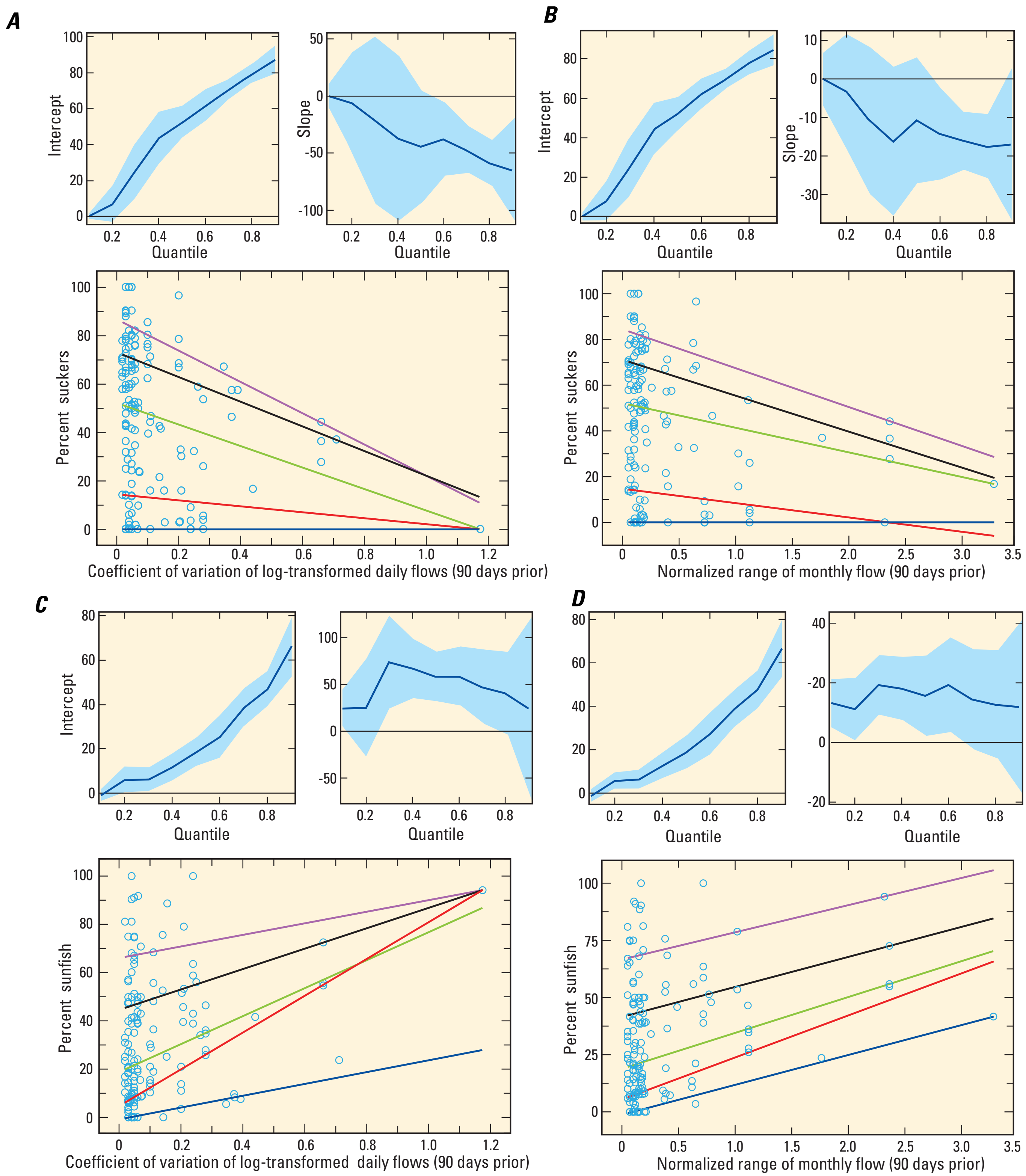


Figure 40. Graphs showing quantile regression lines, intercepts, and slopes of the 0.10, 0.25, 0.50, 0.75, and 0.90 quantiles for significantly correlated 365-day hydrologic and fish metrics for the middle Verde River, Arizona. *A*, percentage of suckers (Catostomidae) and minimum flow divided by the median flow; *B*, percentage of suckers and median annual number of low-flows, and *C*, percentage of sunfish (Centrarchidae) and the minimum monthly streamflow.



EXPLANATION

Quantile

- 0.1
- 0.25
- 0.5
- 0.75
- 0.9

Blue bands represent 95-percent confidence intervals

Figure 41. Graphs showing quantile regression lines, intercepts, and slopes of the 0.10, 0.25, 0.50, 0.75, and 0.90 quantiles for significantly correlated 90-day hydrologic and fish metrics for the middle Verde River, Arizona. *A*, percentage of suckers (Catostomidae) and coefficient of variation of the log-transformed mean daily flows ; *B*, percentage of suckers and the normalized range of monthly flow (NRMF); and *C*, percentage of sunfish (Centrarchidae) and coefficient of variation of the log-transformed mean daily flows; and *D*, percentage of sunfish and NRMF.

are related to morphological characteristics, such as body size and shape, feeding habits, and spawning requirements (Lytle and Poff, 2004). Understanding the microhabitat characteristics required by native species is necessary for understanding how these species will be affected by microhabitat alteration related to changes in streamflow.

Fish Data Collection Methods for Pilot Microhabitat Study

The microhabitat pilot study was conducted in 2012 as part of a collaborative effort between NAU and USGS. Research focused on fish and macroinvertebrate data collection, geomorphology, hydrology, and microhabitat at five locations within the Verde River watershed—three upper-middle Verde River watershed locations (Verde River below Granite Creek, Verde River at Perkinsville, and the Verde River near Paulden gaging station) and two large tributaries (West Clear Creek near Camp Verde gaging station and Oak Creek Near Cornville gaging station) (fig. 25).

Fish data were collected using three methods—(1) electrofishing using a backpack electrofisher, (2) seining, and (3) visual underwater identification while snorkeling. Snorkeling and seining methods were used where conditions permitted. Electrofishing took place in subreaches where microhabitat types had high water velocities or there was too much suspended sediment for visual identification. Electrofishing took place in all five sample reaches but not microhabitat types. Fish were measured for total length using a metric ruler then returned unharmed to the stream away from electrofishing activity once they regained mobility. When snorkeling, fish were identified to species (Reynolds, 1983), and total fish length was estimated using a ruler underwater. The snorkeler dropped a numbered, weighted float where a fish was observed, so that microhabitat measurements could be made later at the location of the identified fish. Pools and runs were snorkeled at West Clear Creek and Oak Creek. Seining was also used as a sampling method to collect fish in deep pools. Seines were used as block nets in the high-velocity riffles at the Paulden and Perkinsville Verde River locations. The specific location where each fish was collected or observed was revisited within 30 minutes following fish identification for microhabitat data collection.

Microhabitat Characterization Methods for Pilot Microhabitat Study

Microhabitat measurements were made at the point where a fish was collected or observed during the snorkeling, electrofishing, and seining surveys, as well as from 50 randomly

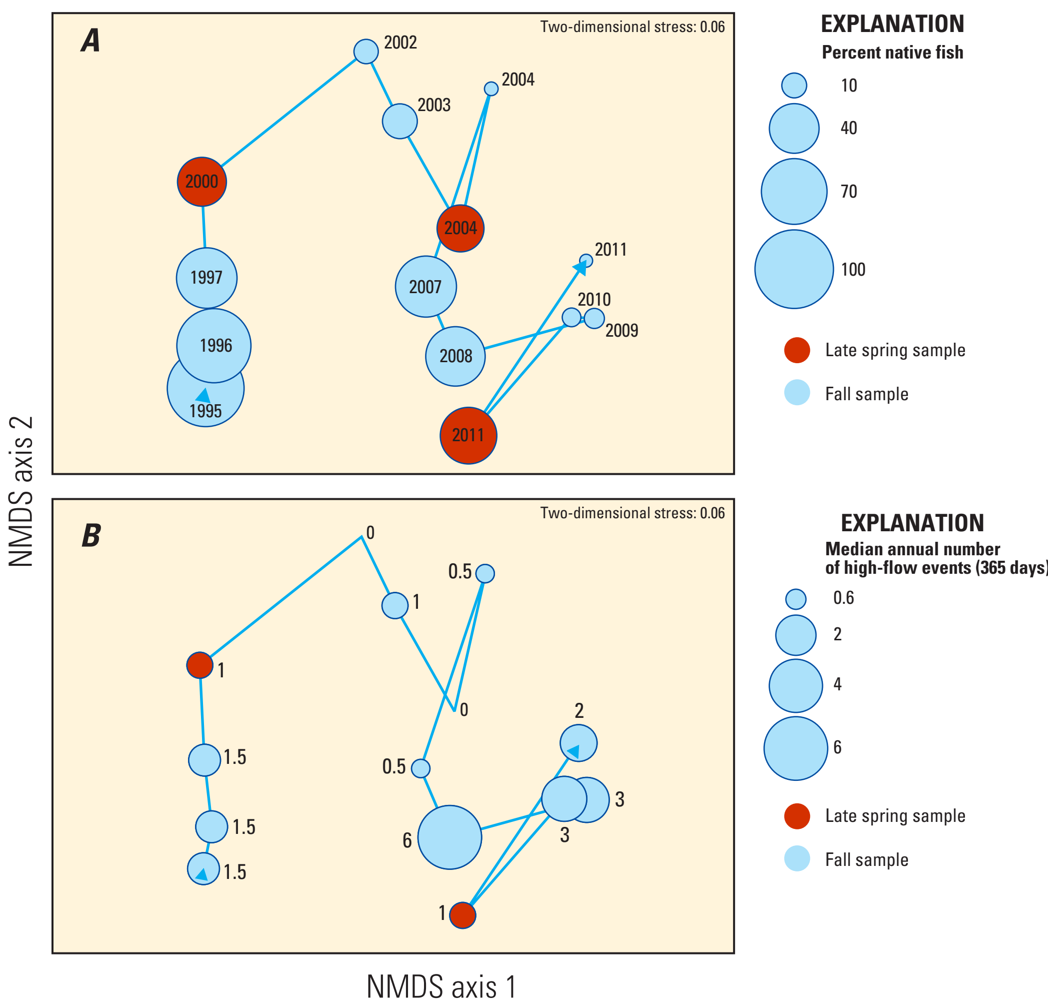


Figure 42. Diagrams showing nonmetric multidimensional scaling (NMDS) ordination plot of assemblages collected from West Clear Creek in 1995–2011 for the middle Verde River watershed, Arizona. *A*, Percent native fish species is superimposed as a bubble plot. *B*, Same years as *A* but median annual number of high-flow events is shown.

selected points selected from a mapped grid of each reach (45 points at Perkinsville) (table 9). Microhabitat data collected in this study included water velocity, water depth, substrate size (*b*-axis) measurements, and riparian canopy cover. A velocity measurement was taken using the six-tenths depth method (Buchanan and Somers, 1969) at the location of each fish, as well as at each of the randomly selected points using a Marsh-McBirney flow meter, Pygmy meter model 6205, or SonTek flow-tracker. Velocities were measured to hundredths of a foot per second and depths to tenths of a foot. River habitat was described using established fluvial geomorphology classifications (Flosi and others, 2010), along with several common monitoring and assessment protocols (Peck and others, 2006; Moulton and others, 2002). Habitat types were classified as pool (wide, deep, slow moving), glide (fast moving, smooth water surface, wide, shallow), run (fast moving, smooth water surface, narrow, deep), riffle (fast moving, steep, shallow, high Froude number, rough water surface), or rapid (rapid and turbulent, surface with intermittent whitewater with breaking waves) (Flosi and others, 2010; Peck and others, 2006; Moulton and others, 2002; Arend, 1999). Along with each velocity and depth measurement, a modified Wolman Pebble Count (Wolman, 1954) was conducted for substrate by measuring the *b*-axis of 15 randomly selected pebbles. Riparian-canopy cover percentages were measured using a concave spherical densiometer. The riparian cover was recorded facing upstream, downstream, at the left and right banks, and then averaged. The presence of woody debris, algae, aquatic vegetation, and undercut banks was recorded to indicate the availability of cover for fish.

To compare how native and non-native species were using aquatic microhabitat, randomized microhabitat measurements were collected as a way to represent the potential available microhabitat. Categories for velocity and depth were developed using quantile divisions in the random data distribution [for example, the 25th, 50th (or median), and 75th percentile]. Substrate categories were established using published ranges for common substrate sizes (Peck and others, 2006) (see table 9 for specific subcategory ranges, descriptors, and quantiles).

Pilot Microhabitat Study Sampling Locations

Similar to the microhabitat categories, subreaches were classified as geomorphic channel units (GCUs)—a quick visual identification of pool, glide, run, riffle, and rapid—to ensure that multiple microhabitat types are represented in a chosen stream reach (fig. 43). Sampling reaches were chosen to have between 5 and 9 alternating GCUs. This was generally achieved using 20 times the stream-bank full width or 40 times the average wetted width at base flow, but not less than 492 ft (150 m) (Moulton and others, 2002; Peck and others, 2006). Total reach length ranged from a minimum of 656 ft (200 m) to a maximum of 1,312 ft (400 m; table 10). However, total reach length did not necessarily correlate with total number of GCUs. For example, in West Clear Creek, there were eight alternating GCUs—run, riffle, glide, riffle, run, riffle, glide, pool. Each GCU was relatively short and the entire sample reach was about 820 ft (250 m). In the Oak Creek reach there were just five alternating GCUs—glide, run, riffle, glide, pool—but the GCUs were considerably longer in length, and the entire sample reach was about 1,148 ft (350 m).

Total Fish Assemblages in Pilot Study

A total of 15 fish species were identified during the 2012 collection period across all sites (table 11; fig. 44*A*). Five were native and 10 were non-native species. The five native species were desert sucker, Sonora sucker, speckled dace, longfin dace, and roundtail chub. The 10 non-native species collected were western mosquito fish, black bullhead, red shiner, rainbow trout, largemouth bass, smallmouth bass, green sunfish, rock bass, common carp, and yellow bullhead. Certain species were only found at specific locations. For example, longfin and speckled dace were only collected at the Perkinsville site, whereas rock bass, black bullhead, and common carp were only collected at Oak Creek. The percentage of non-native species was greater at the Paulden location and at the tributaries, Oak Creek and West Clear Creek (fig. 44*B*). The Verde River below Granite Creek and the Perkinsville sites were most similar relative to percent native fish, but the Verde River below Granite Creek and the Paulden sites were most similar relative to fish assemblage (fig. 44*C*, left to right along NMDS axis 1). The assemblage change from the Verde River below Granite Creek to Perkinsville is related to a reduction in red shiners and an increase in the two dace species.

Microhabitat Characteristics by Pilot Study Sampling Site

The depth, velocity, substrate, and riparian-cover microhabitat characteristics of the fish-microhabitat study sites varied somewhat (figs. 45*A*, *B*, *C*, *D*; table 9), although there was no significant difference in median depth among the five locations. The Verde River below Granite Creek location had the slowest moving water of the five sites and was shallow with the finest substrates (silt and sand) and nearly 40 percent riparian cover. There was little difference overall in the microhabitat characteristics at the Paulden, Perkinsville, and Oak Creek sites, where stream-flow velocity is generally slow to moderate and depth is shallow to very shallow with gravel substrates. However, the waters are faster at Paulden and the riparian cover greater at Oak Creek. The waters at West Clear Creek were of moderate velocity and shallow with gravel and cobble substrates and significant riparian cover.

Median velocities at the Verde River below Granite Creek site (about 0.13 foot per second, ft/s) were significantly slower than all other sites, and at Paulden (1.39 ft/s) they were significantly faster than all other sites except West Clear Creek (fig. 45*A*). There was no statistically significant difference among the median velocities at Perkinsville, Oak Creek, and West

Table 9. Fish microhabitat data-category quantiles and ranges for five middle Verde River, Arizona, study reaches.

[Categories include velocity, depth, median substrate, and percent riparian cover.]

Quantile, in percent	All Stations	Verde River below Granite Creek	Verde River near Paulden (09503700)	Verde River at Perkinsville	Oak Creek near Cornville (09504500)	West Clear Creek near Camp Verde (09505800)
Velocity, in feet per second						
Minimum	0.00	0.00	0.00	0.00	0.00	0.00
10	0.03	0.01	0.20	0.04	0.07	0.07
25	0.10	0.03	0.68	0.09	0.10	0.15
50 (median)	0.52	0.13	1.39	0.38	0.48	0.49
75	1.20	0.59	1.85	0.93	1.09	1.38
90	2.23	0.88	2.30	2.02	1.77	2.78
Maximum	4.13	2.81	4.13	3.73	3.08	3.61
Depth, in feet						
Minimum	0.20	0.20	0.32	0.30	0.20	0.30
10	0.50	0.50	0.65	0.52	0.56	0.45
25	0.75	0.70	0.89	0.75	0.90	0.68
50 (median)	1.10	1.03	1.10	1.00	1.30	0.98
75	1.50	1.99	1.40	1.30	1.70	1.53
90	2.32	2.98	1.79	1.74	2.34	2.64
Maximum	5.00	3.25	3.12	3.00	5.00	3.20
Median substrate, in millimeters						
Minimum	0.00	0.00	0.00	0.00	0.00	0.00
10	0.00	0.00	2.00	0.00	0.00	9.10
25	2.00	0.00	5.00	0.00	2.00	41.25
50 (median)	16.00	2.00	16.00	29.00	20.00	76.00
75	66.00	2.00	43.75	100.00	61.00	114.75
90	104.40	2.00	69.80	118.80	79.00	139.60
Maximum	294.00	40.00	90.00	150.00	90.00	294.00
Riparian cover, in percent						
Minimum	0.0	0.0	0.0	0.0	0.0	0.0
10	0.0	0.0	0.0	0.0	0.0	31.3
25	1.5	11.1	1.5	0.0	0.0	58.8
50 (median)	32.4	41.2	6.7	8.8	55.9	74.3
75	72.1	72.5	17.3	32.4	79.0	96.0
90	96.2	97.0	40.8	53.2	99.9	100.0
Maximum	100.0	100.0	76.5	100.0	100.0	100.0

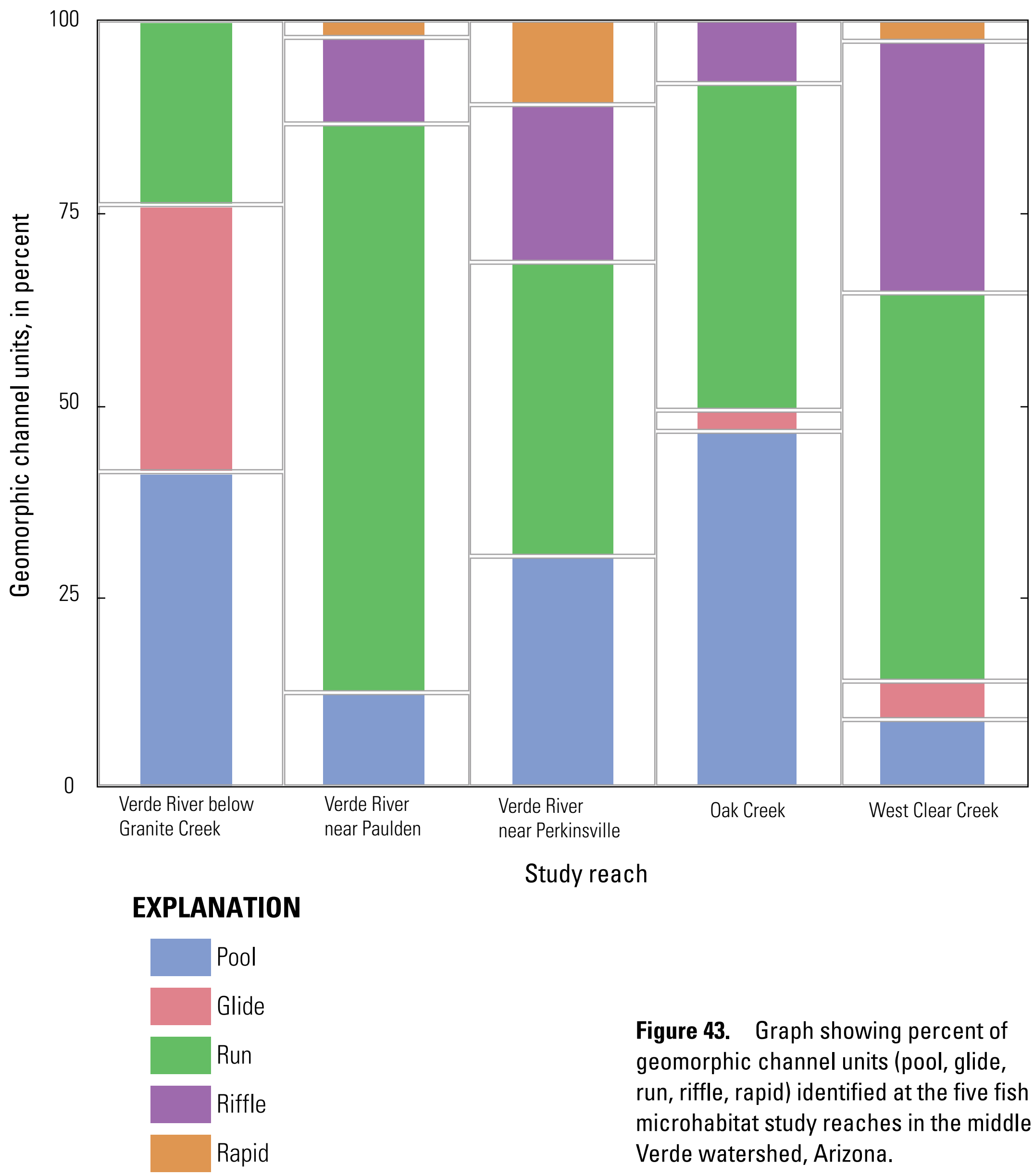


Figure 43. Graph showing percent of geomorphic channel units (pool, glide, run, riffle, rapid) identified at the five fish microhabitat study reaches in the middle Verde watershed, Arizona.

Table 10. Fish-microhabitat study-reach length and geomorphic channel unit sequence for the middle Verde watershed, Arizona.

Study reach	Reach length, in feet	Geomorphic channel unit type present
Verde river below Granite Creek	985	pool, glide, run
Verde River near Paulden (09503700)	1,148	pool, glide, rapid, riffle, run
Verde River at Perkinsville	985	pool, glide, riffle, run
Oak Creek near Cornville (09504500)	985	pool, glide, riffle, run
West Clear Creek near Camp Verde (09505800)	820	pool, glide, rapid, riffle, run

Clear Creek. The single highest instantaneous value (4.13 ft/s) was recorded at Paulden. Oak Creek was the deepest site overall due to deeper pools (median depth 1.3 ft), and West Clear Creek was the shallowest (median depth 0.98 ft) (fig. 45*B*). Minimum depths ranged from 0.20 to 0.32 ft and maximum depths from 3 to 5 ft.

Substrate median size increased from upstream to downstream on the upper-middle Verde River (fig. 45*C*). The Verde River below Granite Creek sample reach was comprised of substrate significantly smaller in size (mainly sand and silt, ≤2 millimeter, mm) than at the other four sites, and at West Clear Creek the substrate was significantly larger than the other sites (76 mm), and included the greatest number of cobbles and boulders. Paulden, Perkinsville, and Oak Creek all had gravel substrates (16 mm, 29 mm, and 22 mm, respectively) and were statistically similar. Median percent riparian cover was significantly greater (p<0.0001–0.0203, Wilcoxon pairwise comparisons) at the Verde River below Granite Creek, Oak Creek, and

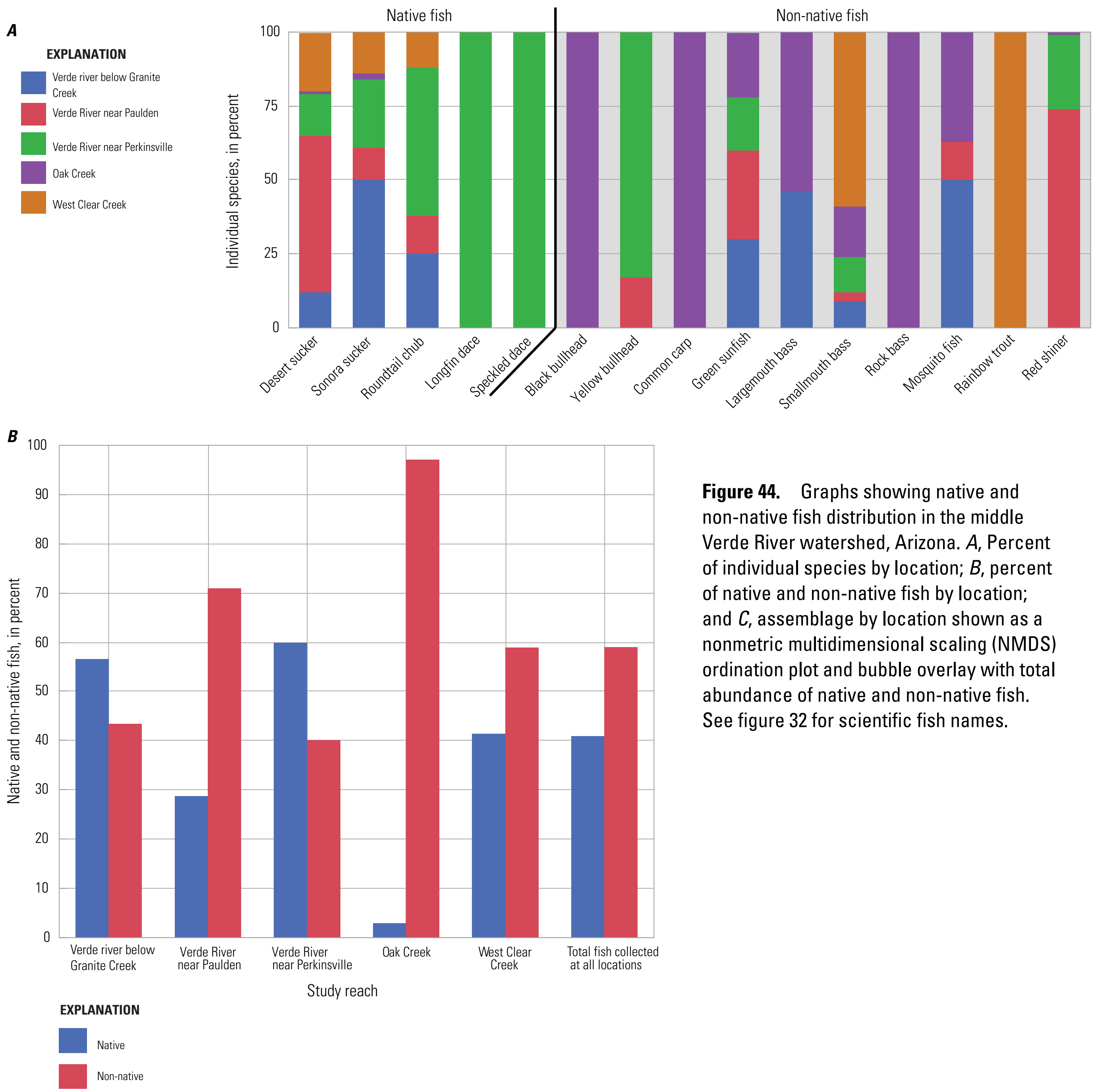


Figure 44. Graphs showing native and non-native fish distribution in the middle Verde River watershed, Arizona. *A*, Percent of individual species by location; *B*, percent of native and non-native fish by location; and *C*, assemblage by location shown as a nonmetric multidimensional scaling (NMDS) ordination plot and bubble overlay with total abundance of native and non-native fish. See figure 32 for scientific fish names.

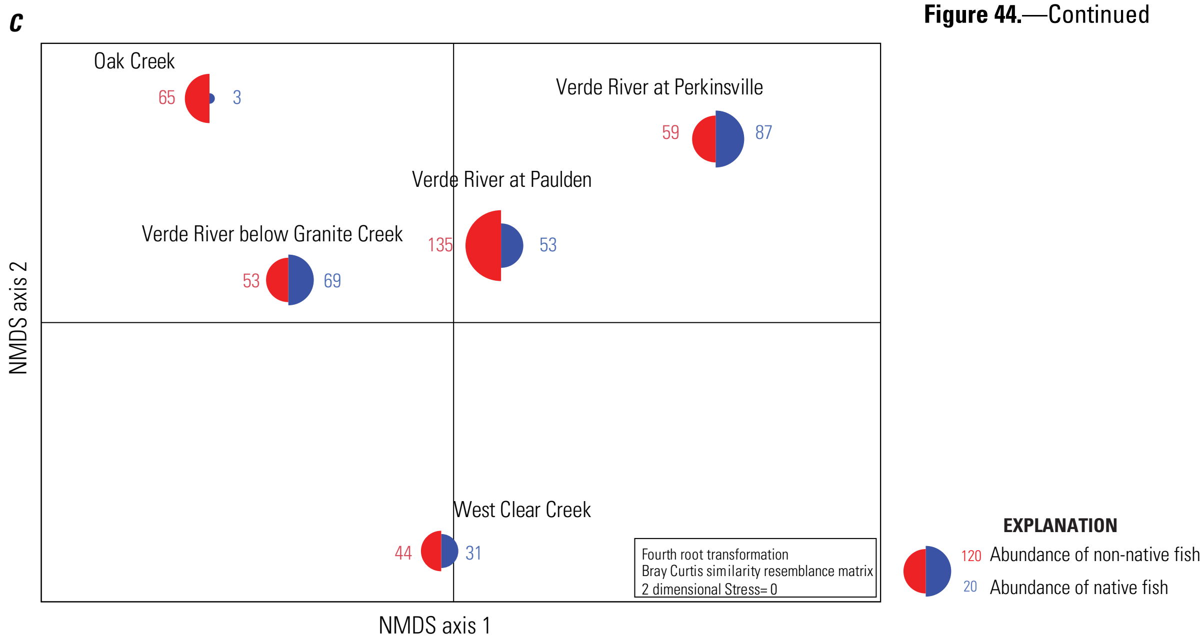


Figure 44.—Continued

Table 11. Number and percent of native and non-native fish species at each sample reach in the middle Verde River watershed, Arizona.

[See figure 32 for scientific fish names. %, percent]

Species	Verde River below Granite Creek	Verde River near Paulden (09503700)	Verde River at Perkinsville	Oak Creek near Cornville (09504500)	West Clear Creek near Camp Verde (09505800)	Total Abundance
			Native			
Desert sucker	9	39	10	1	14	73
Sonora sucker	58	13	26	2	16	115
Roundtail chub	2	1	4	0	1	8
Longfin dace	0	0	27	0	0	27
Speckled dace	0	0	20	0	0	20
			Non-native			
Black bullhead	0	0	0	2	0	2
Yellow bullhead	0	1	5	0	0	6
Common carp	0	0	0	8	0	8
Green sunfish	10	10	6	7	0	33
Largemouth bass	13	0	0	15	0	28
Smallmouth bass	6	2	8	12	41	69
Rock bass	0	0	0	2	0	2
Mosquito fish	24	6	0	18	0	48
Rainbow trout	0	0	0	0	3	3
Red shiner	0	116	40	1	0	157
Total	122	188	146	68	75	599
			Native versus non-native			
Native	69	53	87	3	31	243
Non-native	53	135	59	65	44	356
Percent native	57%	28%	60%	4%	41%	41%
Percent non-native	43%	72%	40%	96%	59%	59%

West Clear Creek sites than at Paulden and Perkinsville due to the narrower stream channels (fig. 45*D*). West Clear Creek had significantly more cover—74 percent—than all of the other sites, whereas Paulden and Perkinsville had just 7 and 9 percent, respectively.

Pilot-Study Fish Characteristics

The pilot-study analysis focused on similarities and differences in habit use by native and non-native fish species. Fish observations were plotted in relation to depth and velocity measurements collected at the respective observation location and visualized using a nonparametric density plot to show a smooth, nonparametric bivariate surface that describes the density of all observations. Velocities measured at the same depths where fish were observed showed the highest density of total fish presence in waters flowing less than 1 ft/s and at depths between 0.5 and 2.0 ft (fig. 46*A*). Native fish presence was most often found in slow, deep waters (1–3 feet and less than 1 ft/s; fig. 46*B*), and non-native fish were found in slow, shallow waters (0.5–2.0 ft and less than 1 ft/s; fig. 46*C*). Several native fish were also found in waters with velocities of about 2 ft/s in very shallow to moderate depths (0.5–1.5 ft). Fewer non-native fish were also collected in this velocity range, but native fish appeared to be found more often and in a wider range of depths.

Median water velocities ranged from about 0.01 ft/s for mosquito fish to 2.3 ft/s for speckled dace (fig. 47*A*). Many of the small fish species (longfin dace, red shiner, and speckled dace) were found in a wider range of velocities. Of the more common fish observed in the pilot study, smallmouth bass, Sonora suckers, and green sunfish were more often found in areas of slower velocities related to deeper waters, and desert suckers and red shiners were found in faster waters (fig. 47*B*). Longfin dace occurred in the shallowest waters (0.2 ft), and common carp occurred in the deepest waters (5 ft). Although their sample size was small, roundtail chub occurred across a wide range of depths. Desert suckers used the largest substrate (median 51 mm) and occurred over the widest range of substrate diameters (2 to 210 mm; fig. 47*C*). Sonora suckers and green sunfish were observed near the smallest median substrates (6 mm). Roundtail chub and red shiners were located near coarse gravel substrates; roundtail chub also used cobbles.

Although some fish, such as smallmouth bass, green sunfish, desert sucker, Sonora sucker, roundtail chub, and longfin dace were observed in a range of velocities and depths, it should be noted that age class was not taken into account in the microhabitat-use analysis for the study. The lengths of the individual fish were measured and used as a coarse proxy to determine the distribution of age class and occurrence in different microhabitat types. Figure 48 shows the significant relations for species where depth or velocity explained a part of the variability between fish length and presence in microhabitat at the location of collection. For the species shown in the linear regression plots, the length and potentially the age of a fish appears to affect the its distribution in relation to depth and velocity. There was general pattern of larger fish (>150 mm), both native and non-native, found in deeper waters (1–3 ft). Velocity relations were less clear. In future studies, the variation attributable to age class should be taken into account.

Native and Non-native Fish Use of Available Microhabitat in Pilot Study

Under the null hypothesis, the preferential use of microhabitat type by fish would be distributed across all depths, velocities, and substrates consistent with microhabitat availability. Although at least some native and non-native fish were found in each velocity, depth, and substrate category, preferential microhabitat use was common (figs. 49, 50, 51; table 12). On a percentage basis (figs. 49*A*, 50*A*, 51*A*), non-native species had a strong preference for slow-moving and deeper water with silt and sand substrate, with a secondary preference for faster moving and very shallow water and a coarse gravel substrate. Native species showed a general preference for moderately fast, moderately deep water over coarse gravel and had no clear secondary preference. The density curves (fig. 49*B*, 50*B*, 51*B*) are a function of the number of items in each of the bins in the figures 49*A*, 50*A*, and 51*A*. This results in a somewhat different shape to the density curve when compared to the quantile histogram, especially for figure 51*B* versus figure 51*A*.

Native fish disproportionately used slow velocity microhabitat (>0.10–0.52 ft/s) compared to the availability of this microhabitat type in the field (fig 49*A*). Moderate velocities (>0.52–1.12 ft/s) accounted for 24 percent of the available microhabitat and 23 percent of the native species, but only 7 percent of the nonnative species were found there. Another 27 percent of the available velocity microhabitat was in standing water (0.10 ft/s or slower), and native fish used that velocity less than 10 percent of the time, whereas non-native fish were found there 43 percent of the time. Standing water therefore accounted for the greatest contrast between microhabitat availability and its use by non-native fish (fig. 49*A*). The median stream velocity associated with the native fish species was significantly faster than the median velocity associated with the non-native fish species and likewise faster than the median velocity found in the available microhabitat. The median stream velocity associated with the non-native fish was significantly slower than that of the available microhabitat (fig. 49*C*). Although non-native species showed strong affinity for standing water and native species showed more of an affinity for slow to moderate velocities, use was different for different native species. For example, Sonora suckers consistently used slower velocities (mainly <1.0 ft/s), whereas speckled dace were more common in faster velocity microhabitat (>2.0 ft/s; fig. 47*A*). Non-native red shiners were found in greater velocities (median=1.75 ft/s) than most other non-native species.

Native species were significantly more likely to be found in deeper water than non-native species (median difference=0.2 ft). Thirty-one percent of non-native and 19 percent

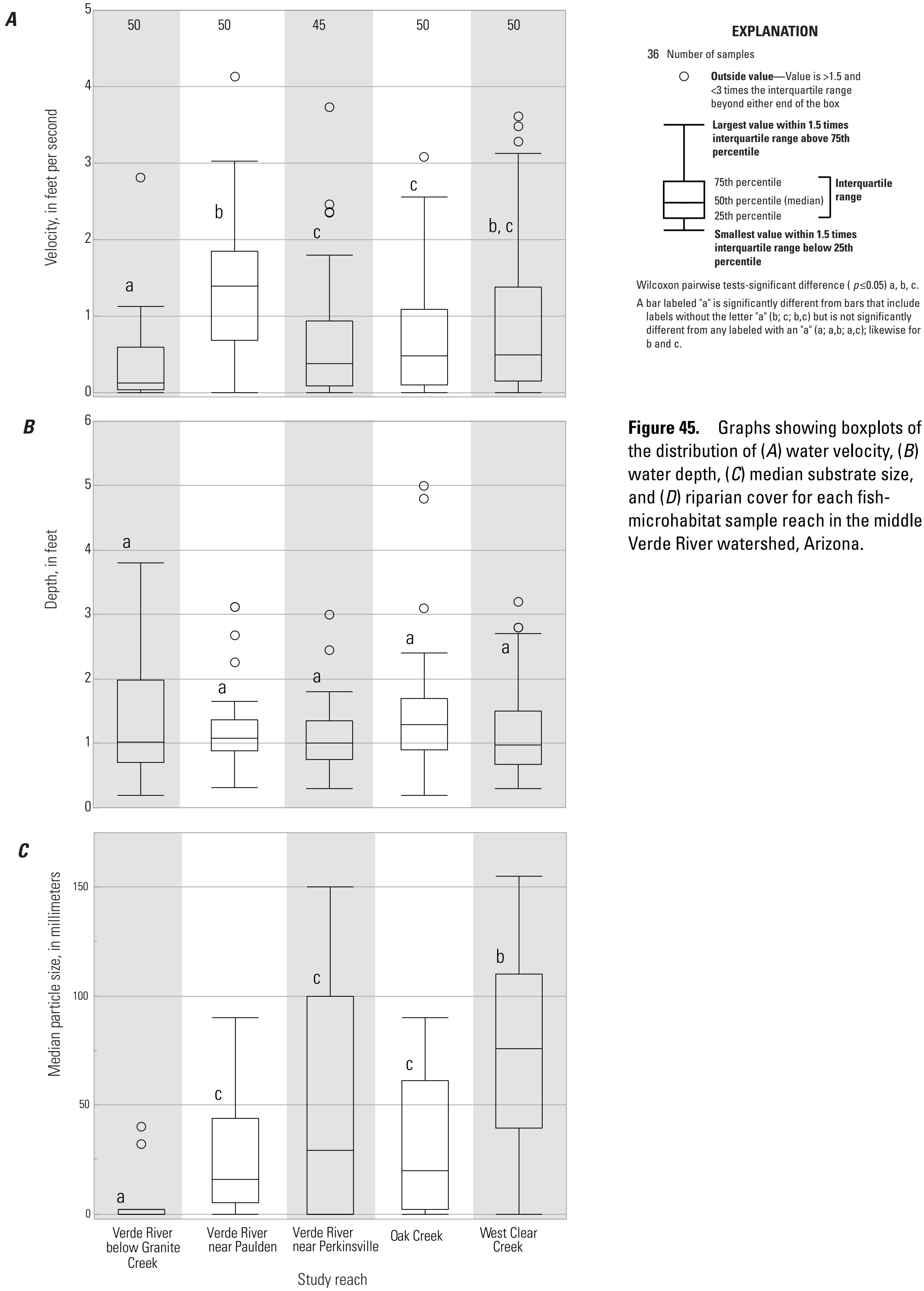
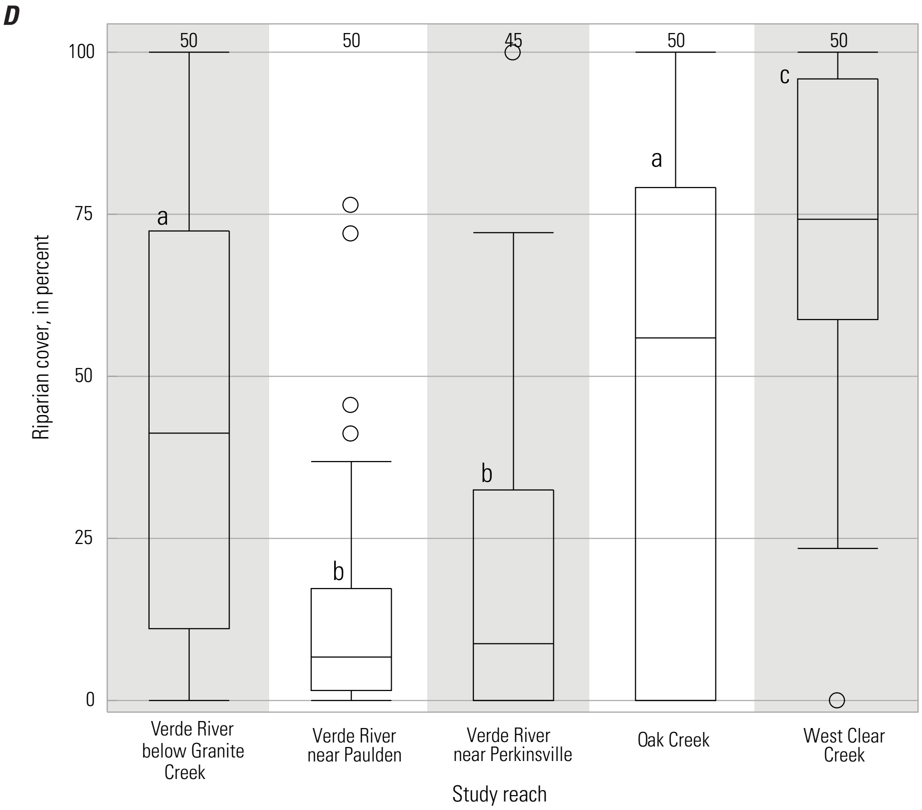


Figure 45. Graphs showing boxplots of the distribution of (*A*) water velocity, (*B*) water depth, (*C*) median substrate size, and (*D*) riparian cover for each fish-microhabitat sample reach in the middle Verde River watershed, Arizona.



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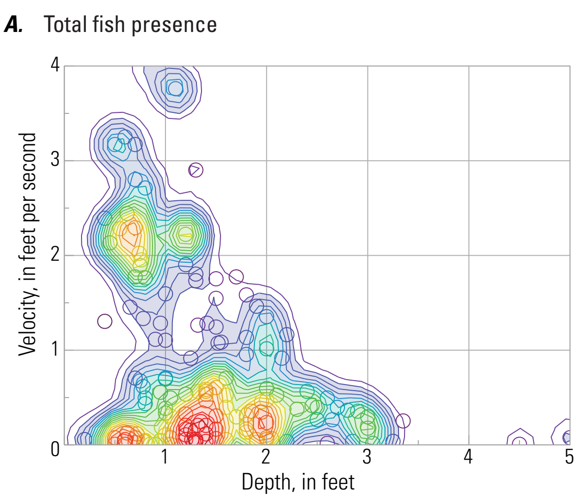
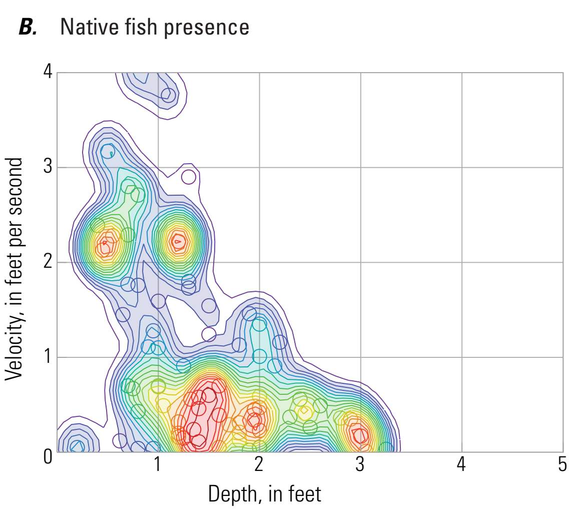


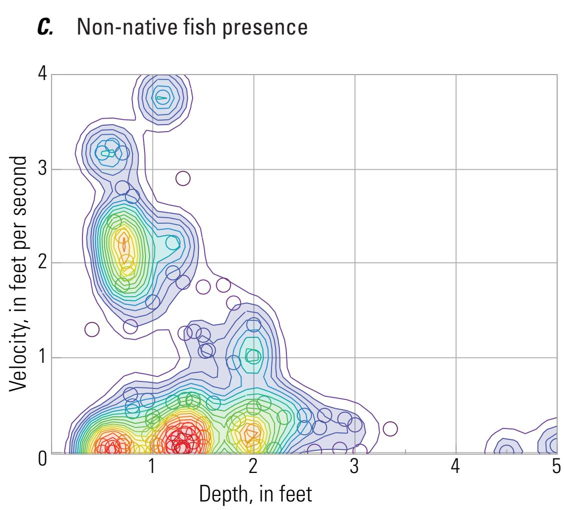
Figure 46. Graphs showing nonparametric density estimates of fish observations for sample reaches in the middle Verde River watershed, Arizona, plotted as a smooth, nonparametric bivariate surface representing the density of observations for a given velocity and depth. *A*, Total fish presence; *B*, native fish presence; *C*, non-native fish presence. The contour lines are quantile contours in 5-percent intervals and colors correspond to the labeled quantile.

EXPLANATION

○ Fish observation

.1 .2 .3 .4 .5 .6 .7 .8 .9 Quantile density contours





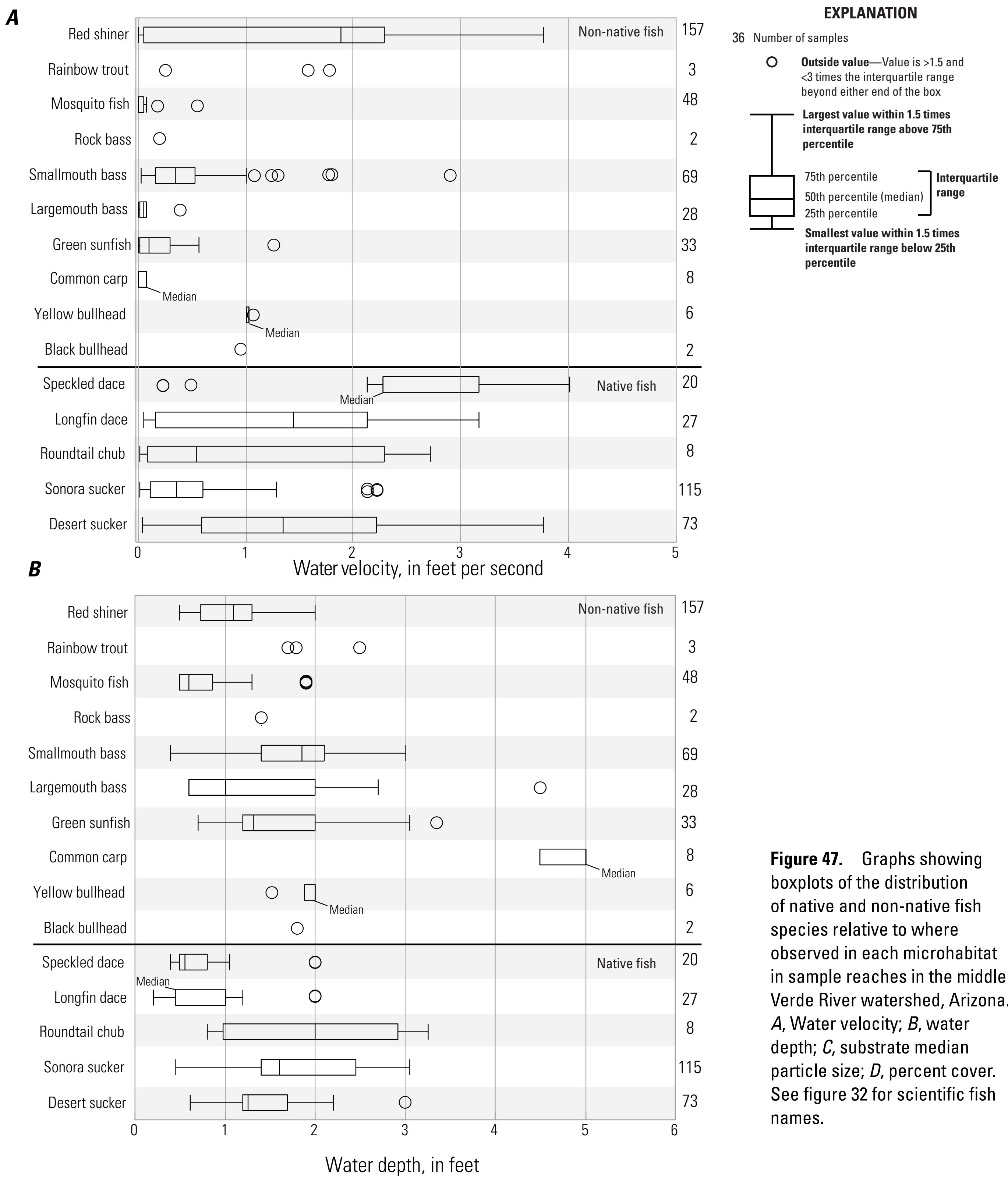
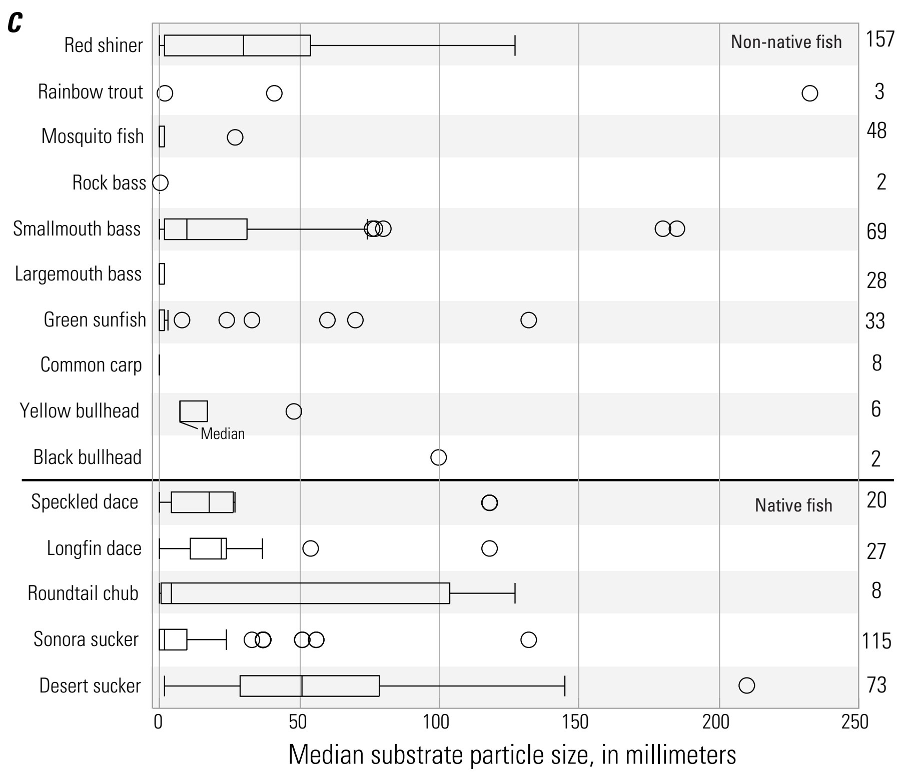


Figure 47. Graphs showing boxplots of the distribution of native and non-native fish species relative to where observed in each microhabitat in sample reaches in the middle Verde River watershed, Arizona. *A*, Water velocity; *B*, water depth; *C*, substrate median particle size; *D*, percent cover. See figure 32 for scientific fish names.



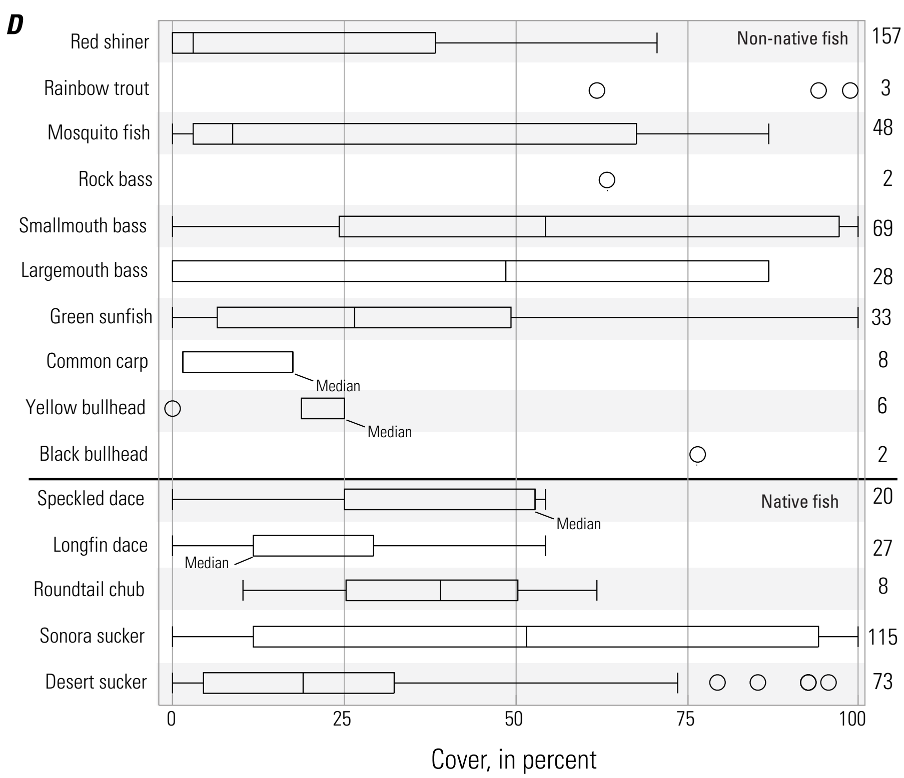


Figure 47.—Continued

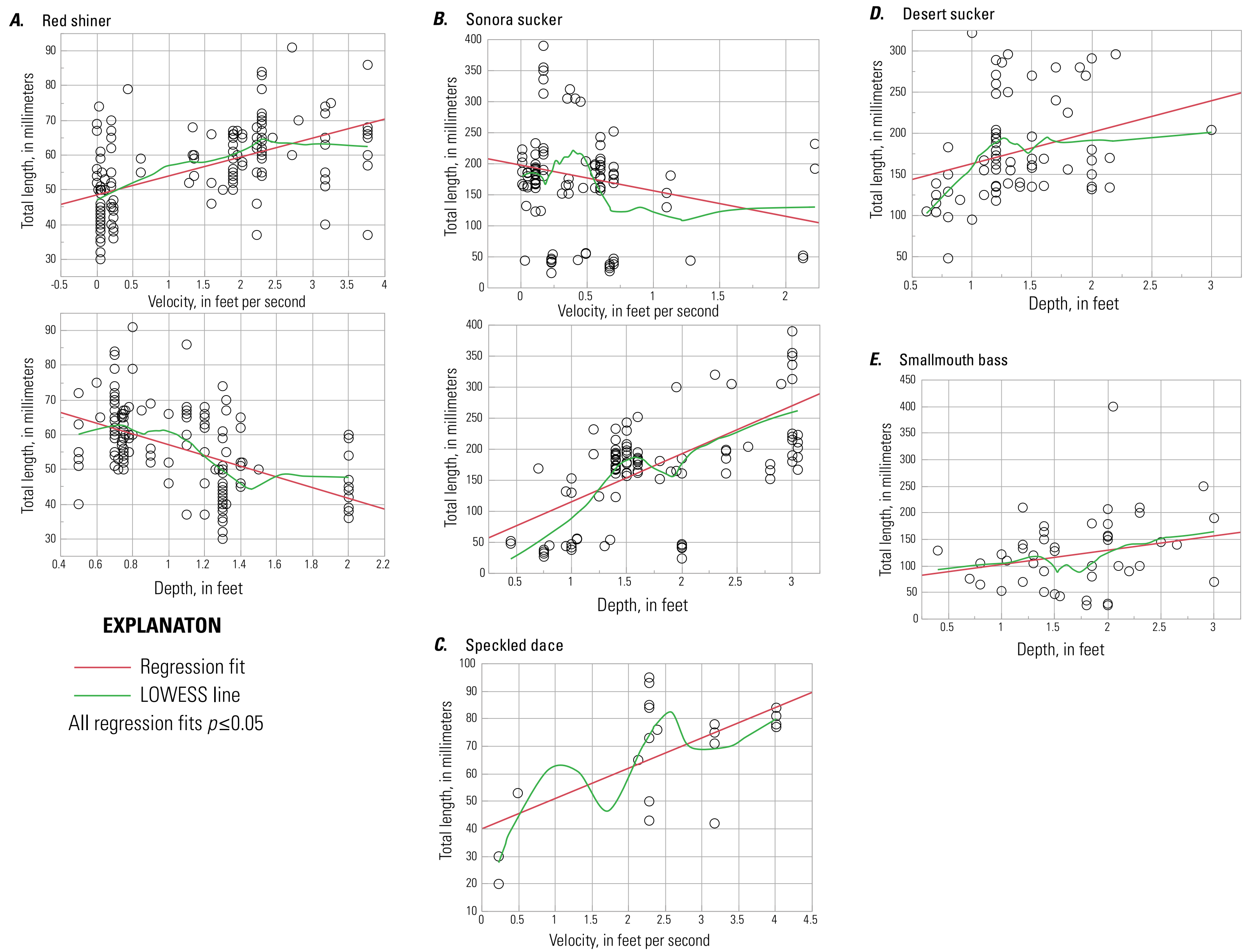


Figure 48. Graphs showing fish length, water depth, and water velocity relations among select fish species from sample reaches in the middle Verde River, Arizona—*A*, red shiner; *B*, Sonora sucker; *C*, speckled dace; *D*, desert sucker; *E*, smallmouth bass. See figure 32 for scientific fish names.

of native fish used water less than 0.75 ft deep. On the other hand, 6 percent of non-native and 16 percent of native fish were found in water deeper than 2.3 ft (fig. 50*A*). It was significantly more likely for the median depth where native species were found to be deeper than the median depth of available microhabitat, whereas for non-native species it was not (fig. 50*C*). Nonetheless, there was much overlap in use between native and non-native fish. For example, both native dace species were observed using shallow depths (less than 1.0 ft) and the two sucker species and roundtail chub were found using depths of about 1.0 to 2.0 ft. This depth use was also common for green sunfish, largemouth bass, and smallmouth bass.

The density curves show that non-native fish used the lowest velocity microhabitat (0 to about 0.3 ft/s), water from 0.5 to 1.5 ft deep, and the smallest diameter substrate (0 to about 25 mm) and the substrate from about 40 to 70 mm diameter. Native fish were found more commonly than non-native fish in about 0.3 to 1.5 ft/s and 2 to 3.1 ft/s velocity microhabitat, water from about 2.3 to 3.6 ft deep, and using substrate of 25 to 40 mm and 100 to 160 mm diameter.

Both native and non-native species used all size substrate (fig. 51*C*). Non-native fish used silt and sand substrates more commonly than native fish and at a rate higher than its availability, whereas native fish were found over gravels at a rate higher than its availability (fig. 51*A*). Both native and non-native fish were less commonly found over cobbles. In general, native fish were significantly more likely to use coarser substrates (in particular, gravels and cobbles) than non-native fish (figs. 51 *A*, *C*). The substrate used by non-native fish was of significantly smaller diameter than that of the available substrate, whereas the substrate used by native fish was not significantly different in diameter from that of the available substrate.

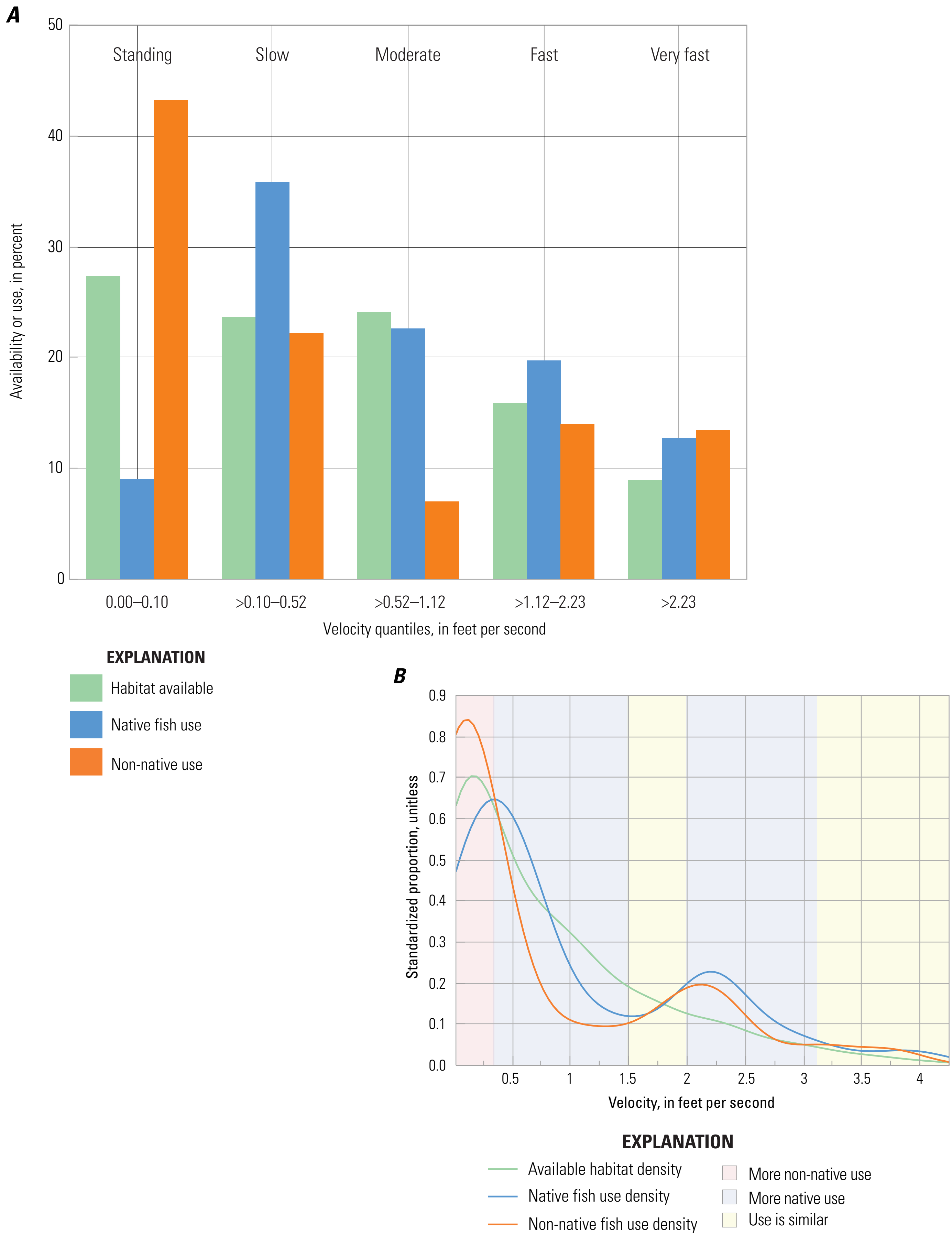
Fish Abundance by Available Microhabitat at Each Pilot-Study Sampling Site

The five sample reaches varied in the relative availability of microhabitat types and in whether the sample reach was dominated by native or non-native species (fig. 44; figs. 52–56). The Verde River below Granite Creek site was characterized by mostly slower moving water (fig. 52*A*). Fifty percent of the microhabitat sampled had water velocities of 0.10 ft/s or less (standing water), and 98 percent of the water velocities sampled were slower than about 1.12 ft/s (standing, slowly, or moderately moving water). More than 80 percent of non-native fish at the Verde River below Granite Creek site were found in standing water, whereas native fish were found in all velocity categories except for very fast. Fifty percent of native species were collected in water faster than 0.52 ft/s. More than 50 percent of available depth at this location consisted of water 1.1 ft deep or less (shallow to very shallow) (fig. 52*B*). Seventy-one percent of non-native species but only 10 percent of native species used these shallow depths. Fifty-two percent of native fish used moderate depth microhabitat (>1.1–1.5 ft).

The Verde below Granite Creek sample reach differed from other reaches by having 96 percent sandy and silty substrate (≤2 mm), and the remaining substrate sampled was coarse gravel (>16–64 mm) (fig. 52*C*). Ninety-four percent of native and 98 percent of non-native fish were found on sand and silt. Overall, native species were more common than non-native species here (fig. 44). Sonora suckers were the predominant native species at this site; desert sucker and roundtail chub were also present (table 11). Non-native fish were dominated by western mosquito fish. The larger predators and competitors—largemouth bass, smallmouth bass, and green sunfish—were also found there but in lower numbers (table 11). Overall, non-native species at the Verde below Granite Creek site preferred very shallow, standing water, but otherwise tended to use microhabitat as it was available. Native fish species tended to use somewhat more specific microhabitats (fig. 52).

The Verde River at Paulden sampling reach was predominantly middle-range in all microhabitat categories—66 percent of microhabitat had moderate or fast water, more than 65 percent had shallow or moderate depths, and 70 percent had fine or coarse gravel. Both native and non-native species used some microhabitat types preferentially. For example, only 4 percent of non-native fish were found in moderate-velocity water, whereas 39 percent were found in standing water, although standing water made up just 8 percent of the available microhabitat. Sixty percent of native fish were found in the fastest-moving waters (>1.12 ft/s), which made up 58 percent of the available microhabitat (fig. 53*A*). Twenty-three percent of the native fish species were found in very deep water, and 2 percent of the non-native fish were found in very deep water. Six percent of the available microhabitat was very deep. On the other hand, 74 percent of non-native fish were found in shallow or moderate depths, and 58 percent of native fish were found in this same middle range (fig. 53*B*). Silt and sand accounted for 18 percent of the available microhabitat. More than 52 percent of the non-native species were found over silt or sand, and 6 percent of native fish were found over silt or sand. Only 1 percent of non-native fish used fine gravel microhabitat at Paulden, even though it accounted for 32 percent of the available microhabitat (fig. 53*C*). Five species of non-native fish were collected in Paulden, constituting 72 percent of the fish collected at that site. Red shiner was the dominant species. Green sunfish, western mosquito fish, smallmouth bass, and yellow bullhead were also collected there. Thirty-nine desert sucker, 13 Sonora sucker, and 1 roundtail chub were the native species observed at Paulden (table 11).

The Verde River at Perkinsville sample reach had predominantly lower water velocities (60 percent slow or standing), shallower depths (82 percent moderate, shallow, and very shallow), and a mix of primarily silt (33 percent) and cobbles (40 percent), plus 16 percent coarse gravel. Non-native species preferred faster water at this site. Fifty-four percent of the non-native fish were found in very fast water (>2.23 ft/s), although the very fast velocity areas made up only 9 percent of the available microhabitat at Perkinsville. Native


A
Standing
Slow
Moderate
Fast
Very fast
50
40
30
20
10
0
Availability or use, in percent
0.00–0.10
>0.10–0.52
>0.52–1.12
>1.12–2.23
>2.23
Velocity quantiles, in feet per second
EXPLANATION
Habitat available
Native fish use
Non-native use
B
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
Standardized proportion, unitless
0.5
1
1.5
2
2.5
3
3.5
4
Velocity, in feet per second
EXPLANATION
Available habitat density
Native fish use density
Non-native fish use density
More non-native use
More native use
Use is similar

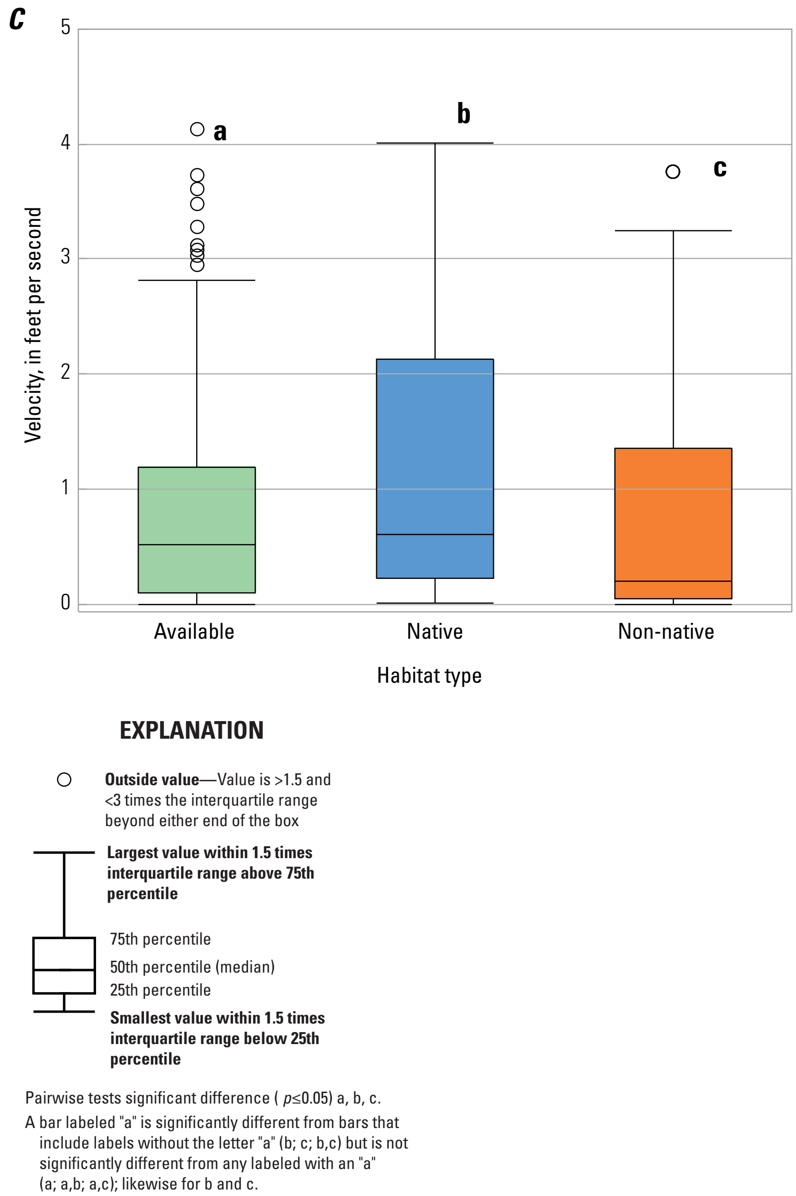
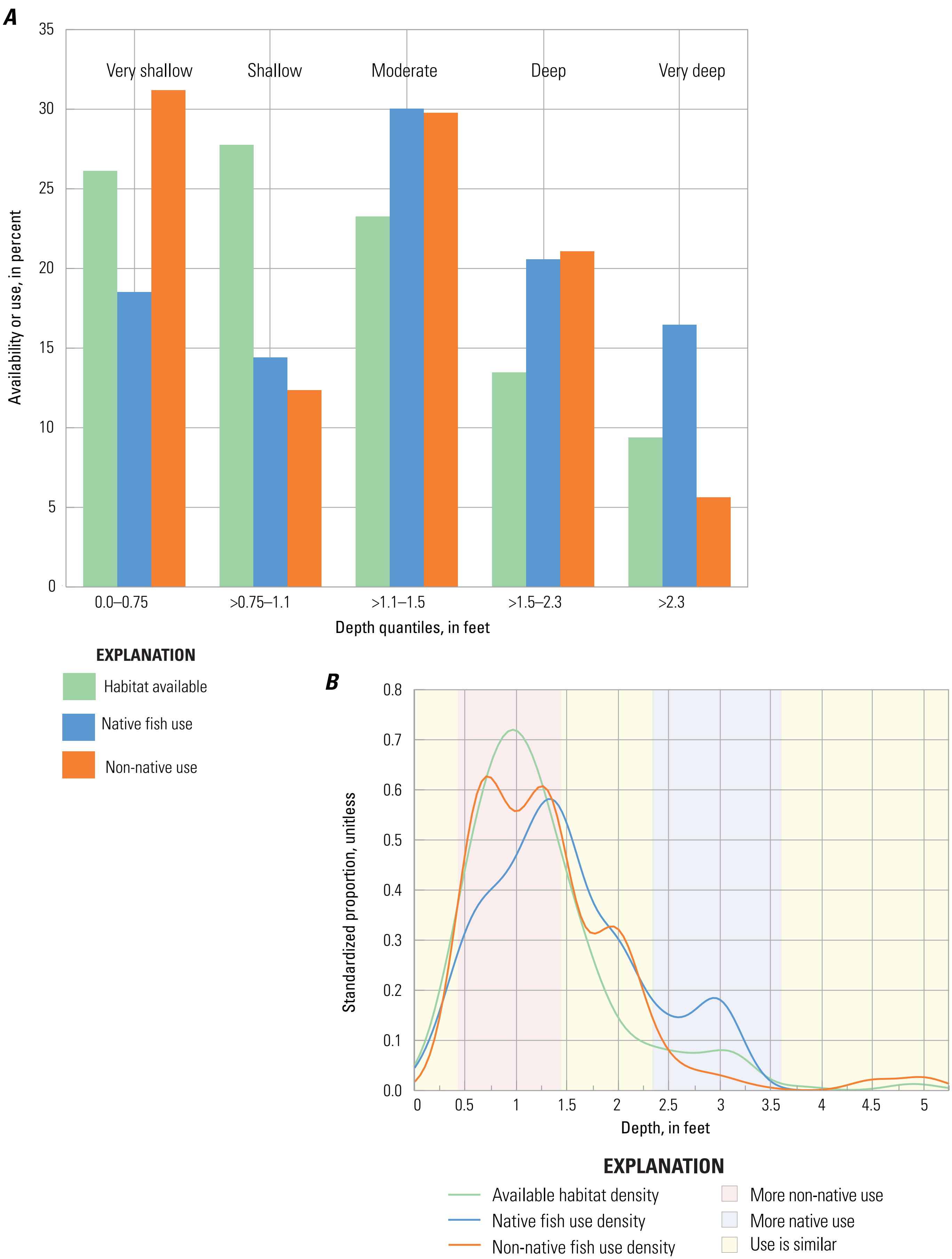


Figure 49. Graphs of available velocity and velocity use by native and non-native fish in sample reaches in the middle Verde River and two tributary streams, Arizona. *A*, Velocity microhabitat quantiles by percent availability or use; *B*, velocity microhabitat availability and native and non-native fish use as a continuous standardized proportion; and *C*, boxplots of velocity microhabitat availability and native and non-native fish median use.


A
35
30
25
20
15
10
5
0
Very shallow
Shallow
Moderate
Deep
Very deep
Availability or use, in percent
0.0–0.75
>0.75–1.1
>1.1–1.5
>1.5–2.3
>2.3
Depth quantiles, in feet
EXPLANATION
Habitat available
Native fish use
Non-native use
B
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
Standardized proportion, unitless
0
0.5
1
1.5
2
2.5
3
3.5
4
4.5
5
Depth, in feet
EXPLANATION
Available habitat density
Native fish use density
Non-native fish use density
More non-native use
More native use
Use is similar

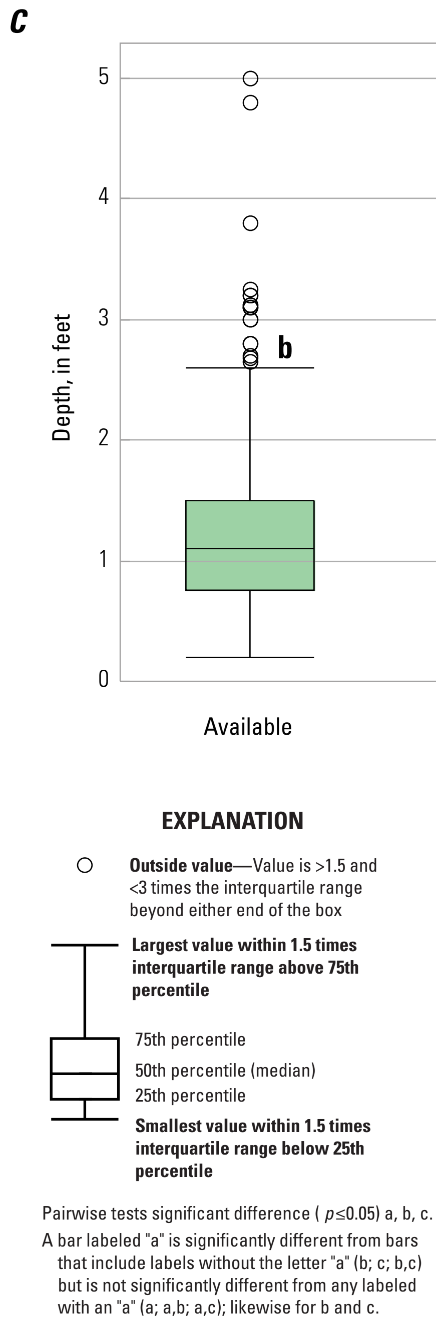
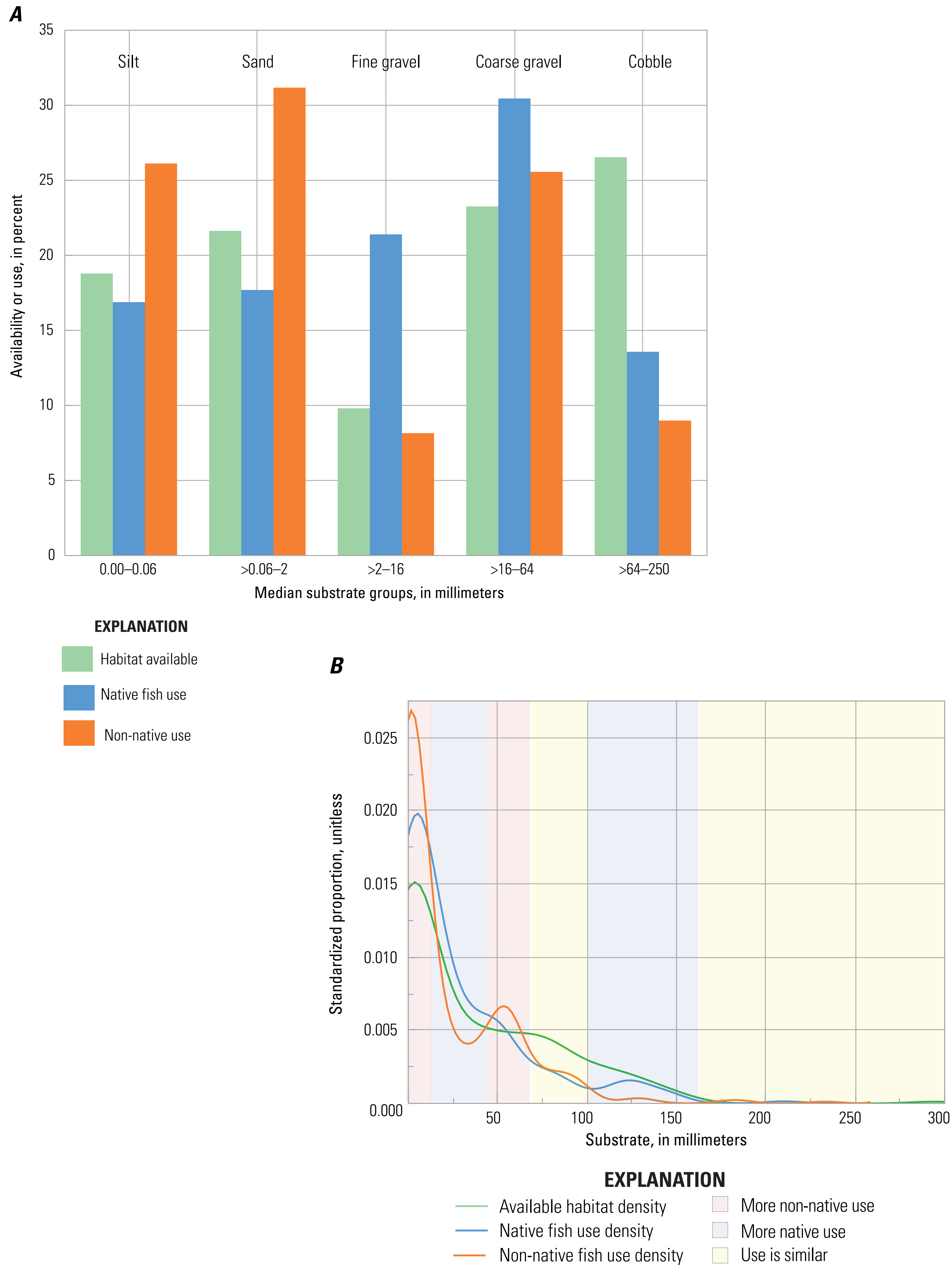


Figure 50. Graphs of available depth and depth use by native and non-native fish in sample reaches in the middle Verde River and two tributary streams, Arizona. *A*, Depth-habitat quantiles by percent availability or use; *B*, depth microhabitat availability and native and non-native fish use as a continuous standardized proportion; and *C*, boxplots of depth microhabitat availability and native and non-native fish median use.


A
Silt
Sand
Fine gravel
Coarse gravel
Cobble
Availability or use, in percent
35
30
25
20
15
10
5
0
0.00–0.06
>0.06–2
>2–16
>16–64
>64–250
Median substrate groups, in millimeters
EXPLANATION
Habitat available
Native fish use
Non-native use
B
Standardized proportion, unitless
0.025
0.020
0.015
0.010
0.005
0.000
50
100
150
200
250
300
Substrate, in millimeters
EXPLANATION
Available habitat density
Native fish use density
Non-native fish use density
More non-native use
More native use
Use is similar

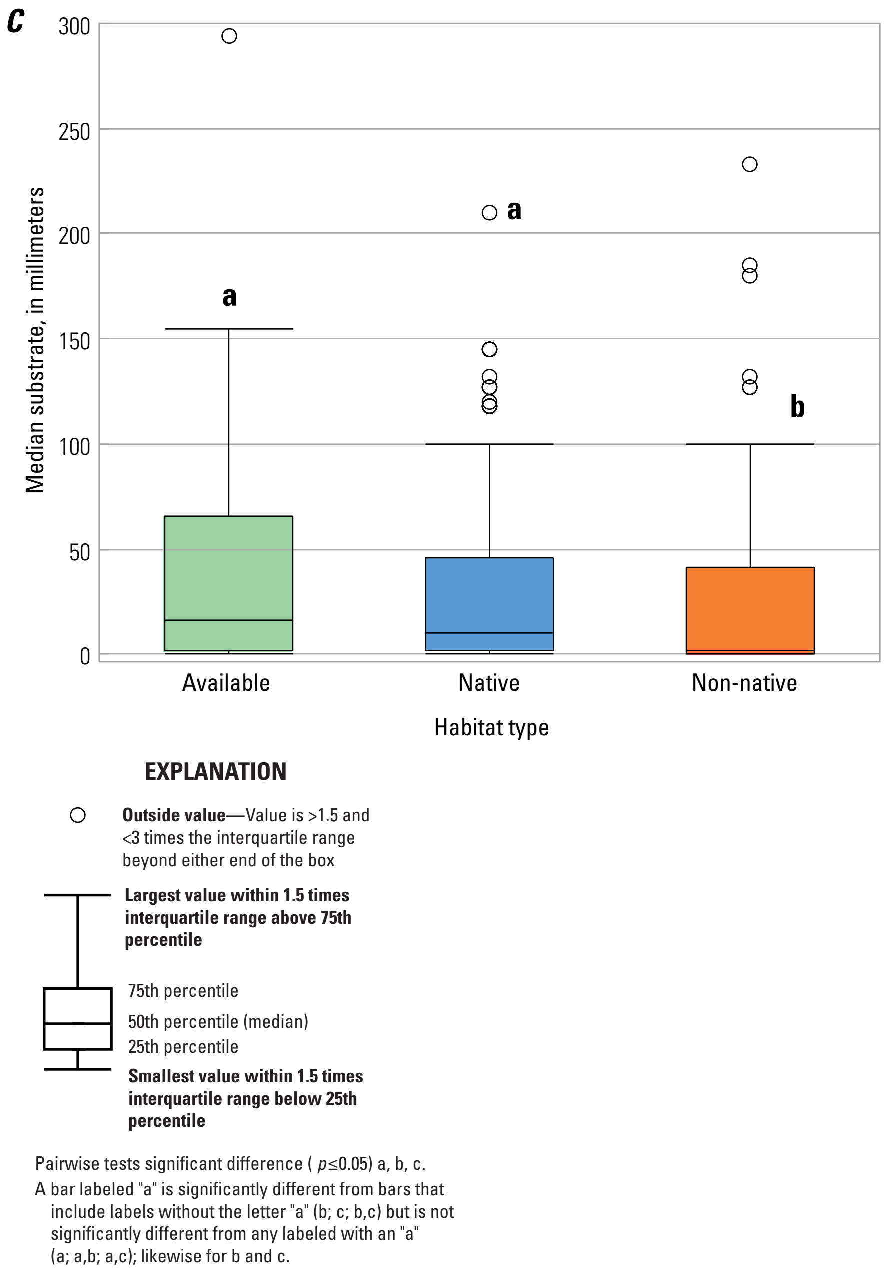


Figure 51. Graphs of available substrate and substrate use by native and non-native fish in sample reaches in the middle Verde River and two tributary streams, Arizona. *A*, Substrate-habitat groups by percent availability or use; *B*, substrate microhabitat availability and native and non-native fish use as a continuous standardized proportion; and *C*, boxplots of substrate microhabitat availability and native and non-native fish median use.

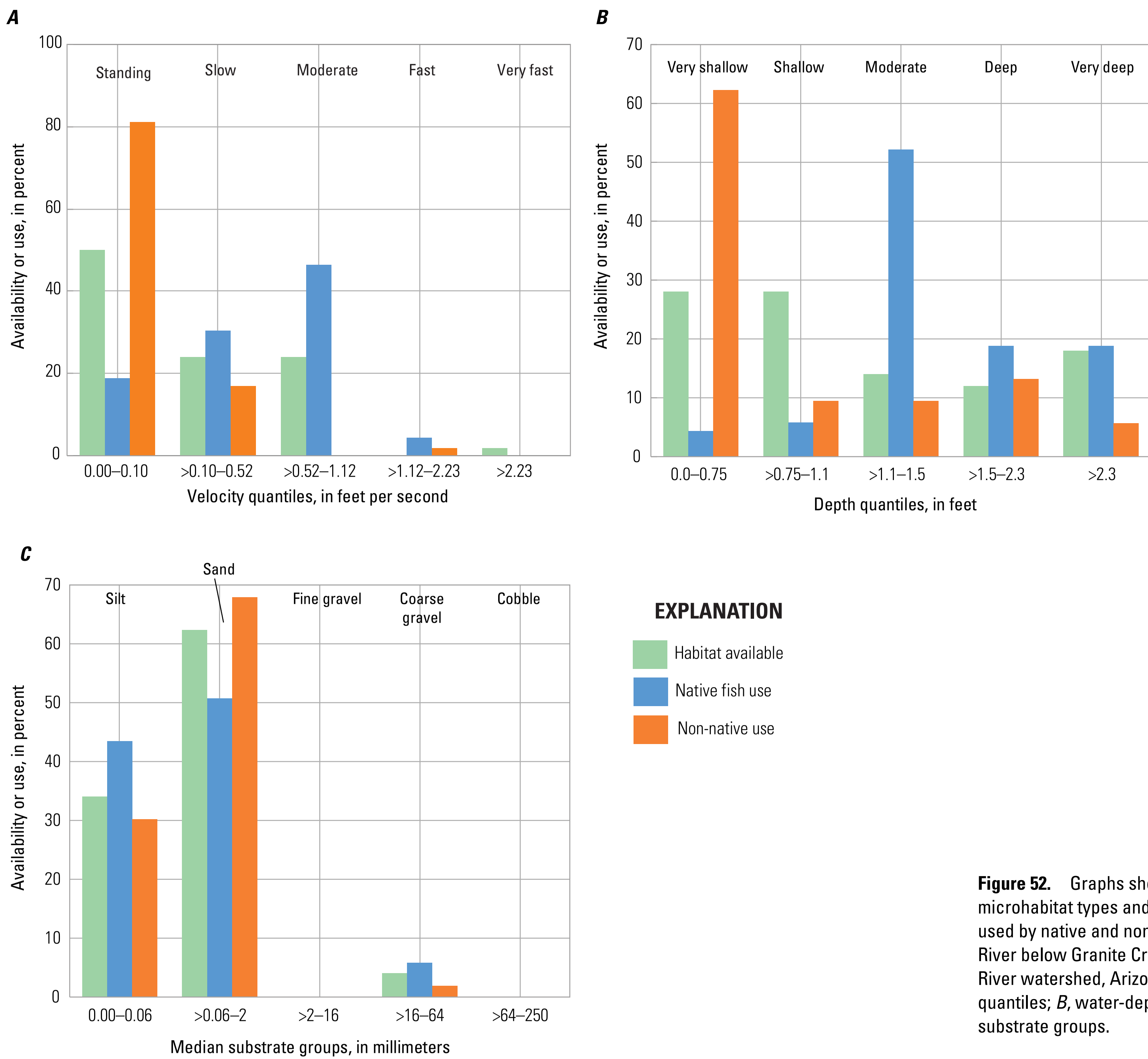


Figure 52. Graphs showing both available microhabitat types and microhabitat types used by native and non-native fish at Verde River below Granite Creek in the middle Verde River watershed, Arizona—*A*, water-velocity quantiles; *B*, water-depth quantiles; and *C*, substrate groups.

fish species were relatively evenly distributed from slow to very fast microhabitats (>0.10 ft/s). Standing water made up 31 percent of the microhabitat but was used by 2 percent of non-native and 6 percent of native fish (fig. 54*A*). Both non-native and native fish preferred both the very shallowest and the deep water at Perkinsville, whereas available microhabitat was predominantly very shallow. In particular, deep and very deep water (>1.5 ft) accounted for 17 percent of the available microhabitat; 25 percent of native fish species and 39 percent of non-native fish species were found in deep water (>1.5–2.3 ft). However, no fish were observed in very deep water (fig. 54*B*). Seventy-two percent of both native and non-native fish were collected over gravels, which accounted for 23 percent of the available microhabitat. Thirty-three percent of the available microhabitat was silt, and 11 percent of native fish species were found there. Less than 5 percent of the available microhabitat at Perkinsville was sand, and no native fish and just a few non-native fish were found there (fig. 54*C*). All five species of native fish observed during this study were found at the Perkinsville location; 60 percent of fish observed were native (fig. 44). This was the only site where speckled dace and longfin dace were found. Red shiners were the most abundant non-native fish species (40 individuals), followed by smallmouth bass, green sunfish, and yellow bullhead.

The fish assemblage, microhabitat type, and physical surroundings at Oak Creek were quite different from the other sample reaches. Oak Creek had the highest proportion of non-native species (96 percent) of all sample reaches (fig. 44). The Oak Creek sample reach was comprised of only 3 individual native fish—one desert sucker and two Sonora suckers. Non-native species collected here but not in any other stream reach included common carp, rock bass, and black bullhead. The only non-native species found elsewhere in this study but not at the Oak Creek site were rainbow trout and yellow bullhead (table 11). Oak Creek was characterized by a broad range of velocity microhabitats and variable depths and substrate. The native fish were collected in slowly flowing, moderately deep or very deep water over coarse gravel and cobble. Non-native fish preferred standing water (83 percent) and silt substrate (74 percent) in Oak Creek, but used all depths as available (figs. 55*A*–*C*).

Thirty-four percent of velocity microhabitat in West Clear Creek was slow-moving water (fig. 56A), and both native and non-native fish disproportionately preferred this slow-velocity flow (77 and 66 percent, respectively). The percent of available water depths declined steadily with increasing depth, from very shallow (36 percent) to very deep (8 percent). Non-native fish preferred depths that were moderate to deep in West Clear Creek (82 percent), even though this represented only 32 percent of the available microhabitat. Native fish preferred deep and very deep water (71 percent), which represented even less of the available microhabitat (22 percent). Thirty-six percent of the available microhabitat was very shallow, but non-native fish used this only 2 percent of the time, and native fish did not use it at all (fig. 56*B*).

West Clear Creek had the greatest median substrate size of all sample reaches (fig 45*C*), with 26 percent of the substrate coarse gravel and 60 percent cobble (fig. 56*C*). Nonetheless, non-native fish were most commonly found over fine-gravel (36 percent), and native fish coarse gravel (42 percent). Silt and sand were the least common microhabitat at 4 percent each. However, neither native nor non-native fish were found over silt. West Clear Creek had more non-native fish (59) than native fish (41) (fig. 44). The native-fish assemblage was comprised of the two sucker species and one individual roundtail chub; the non-native fish were dominated by smallmouth bass plus three rainbow trout.

Non-native fish were most abundant at the Verde River at Paulden study reach by number (135 individuals) and at Oak Creek by percent (96 percent). Oak Creek also had the greatest variety of non-native species (8 total). The study reach with the fewest non-native species by number was West Clear Creek (44 individuals). The greatest abundance of native fish was at Perkinsville by both number and percent (87 individuals, 60 percent of fish collected). Perkinsville also had the largest number of native species (5 species). The fewest individual native fish were collected at Oak Creek (3 suckers), and this was also the site with the fewest number of native species (2 species).

Oak Creek is different from the other sites in that only 3 individual native fish were present and 8 of the 10 non-native fish species collected during the study occurred there (table 11, fig. 44). The only non-native fish species not collected there were yellow bullhead and rainbow trout. This site was also heavily impacted by human use. Part of the stream was diverted, situated near a hatchery, stocked with brown, brook, and rainbow trout (although none of these species were found while sampling Oak Creek), and located next to a recreational vehicle park and campgrounds. The stocking of non-native sport fish and the presence of predatory, competitive non-native species not found in other sample reaches could be affecting native fish presence in Oak Creek (Minckley and Douglas, 1991; Rinne and Janisch, 1995).

Macroinvertebrates

Benthic macroinvertebrates live on streambed substrates, such as cobble, gravel, and sand. Macroinvertebrates play a crucial role in the food web of a stream ecosystem by acting as an intermediary between primary producers and vertebrates. Because macroinvertebrates are relatively sessile and live extended life cycles (a year or more), the macroinvertabrate assemblage is often a function of past stream conditions (Peck and others, 2006). As such, macroinvertebrate assemblages are useful in assessing the status or trend of the ecological condition of a stream. In addition, their function or role in the larger community can be used as a quantitative measure to assess ecological conditions and relate to other stream characteristics.

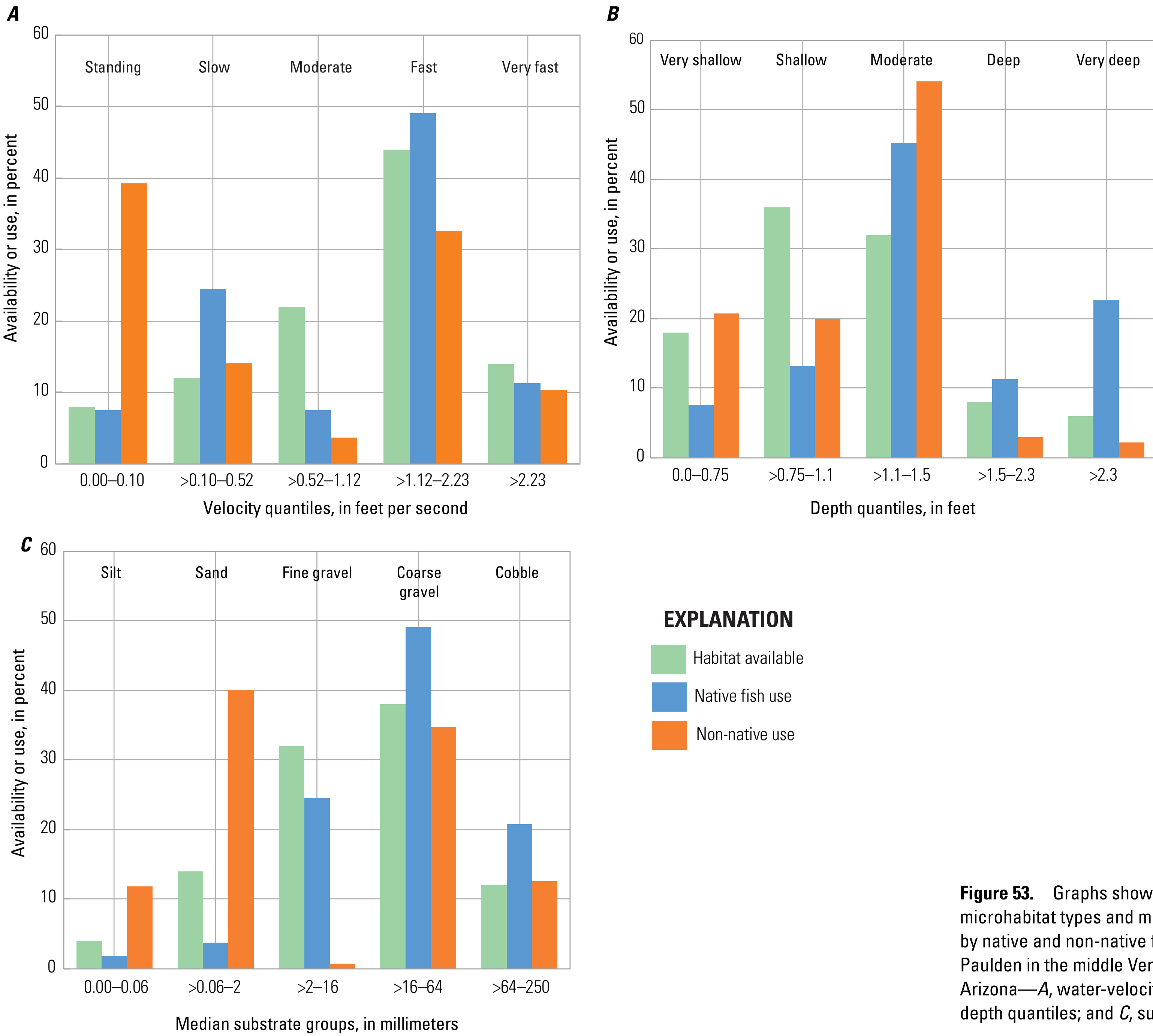


Figure 53. Graphs showing both available microhabitat types and microhabitat types used by native and non-native fish at Verde River at Paulden in the middle Verde River watershed, Arizona—*A*, water-velocity quantiles; *B*, water-depth quantiles; and *C*, substrate groups.

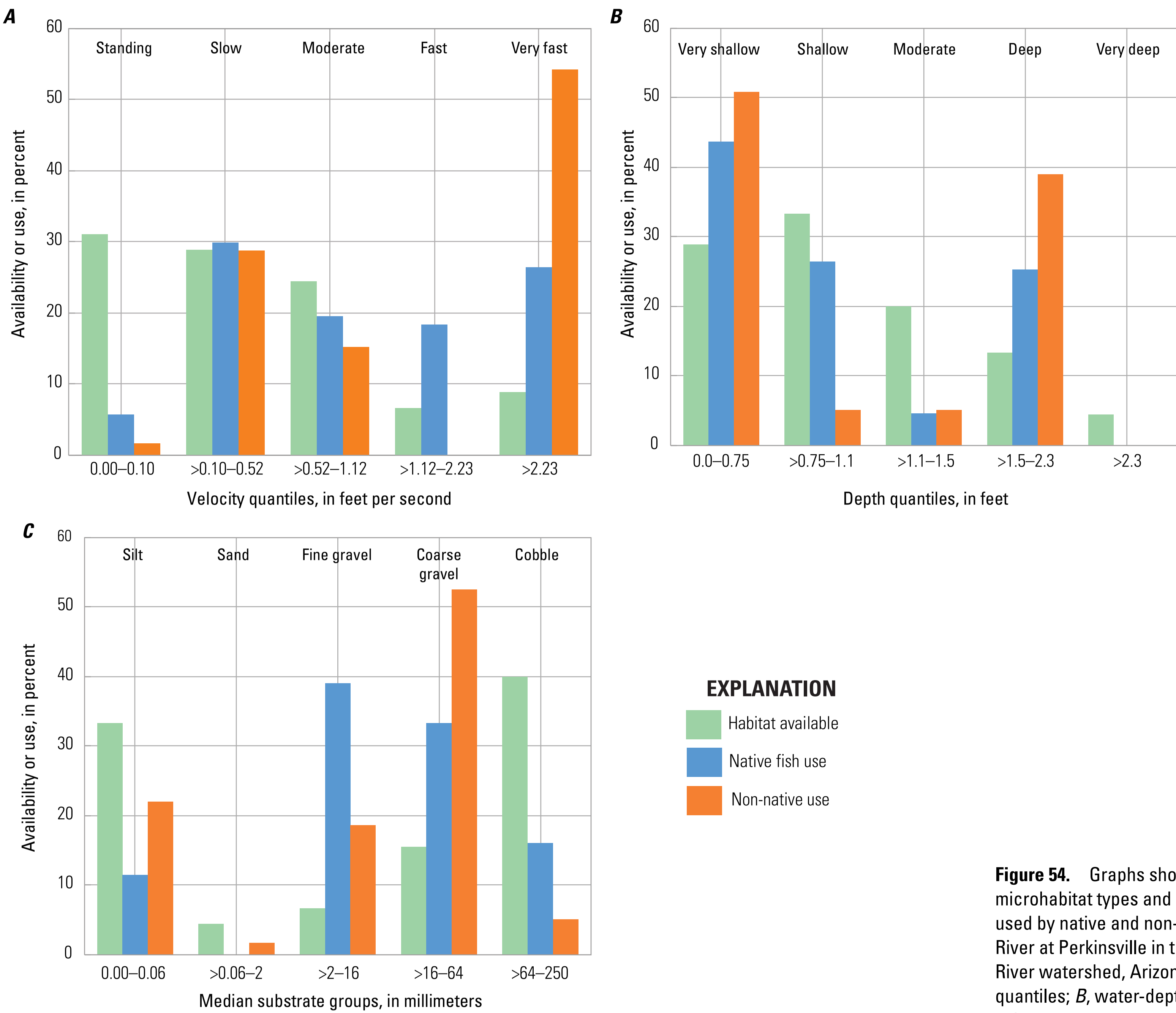


Figure 54. Graphs showing both available microhabitat types and microhabitat types used by native and non-native fish at Verde River at Perkinsville in the middle Verde River watershed, Arizona—*A*, water-velocity quantiles; *B*, water-depth quantiles; and *C*, substrate groups.

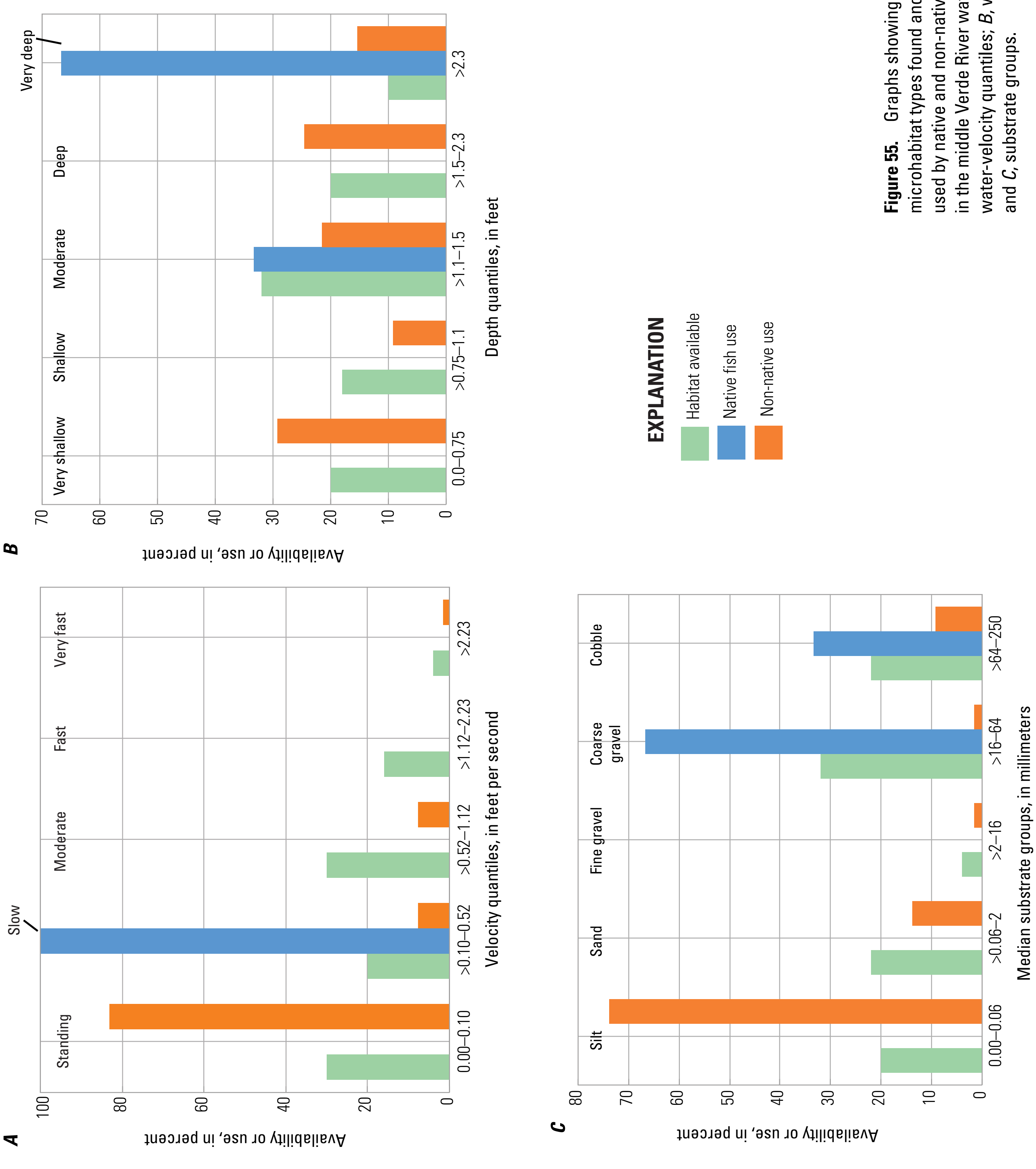


Figure 55. Graphs showing both available microhabitat types found and microhabitat types used by native and non-native fish at Oak Creek in the middle Verde River watershed, Arizona—*A*, water-velocity quantiles; *B*, water-depth quantiles; and *C*, substrate groups.

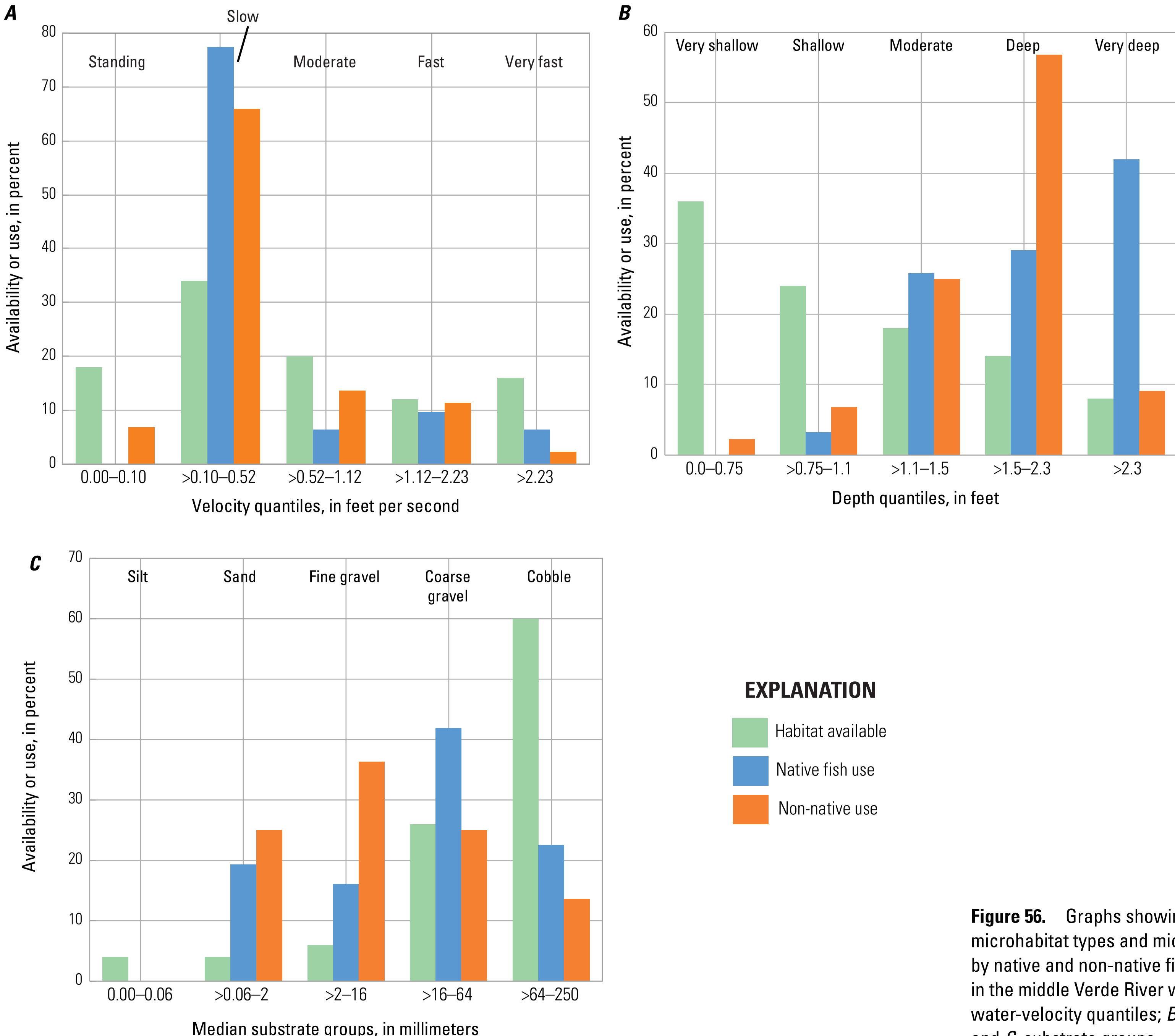


Figure 56. Graphs showing both available microhabitat types and microhabitat types used by native and non-native fish at West Clear Creek in the middle Verde River watershed, Arizona—*A*, water-velocity quantiles; *B*, water-depth quantiles; and *C*, substrate groups.

Macroinvertebrate Metrics

Macroinvertebrate communities are useful indicators of water quality because of their rapid and consistent response to changes in physical and chemical characteristics of streamflow. However, detecting a population response to any one variable can be difficult because many environmental factors can limit population size. Several environmental factors acting in concert can elicit a unique response making the task of identifying a single responsible factor very difficult. Thus, variables in addition to streamflow should be considered when trying to understand macroinvertebrate relations.

Macroinvertebrate communities have been shown to be different depending on the time of year they are sampled (Spindler, 2001). A given sampling time is referred to as an index period and typically spans a season, such as fall or spring (fig. 57). Samples collected following high flow events may be nonrepresentative of stable macroinvertebrate populations because of the slow recovery of some species and (or) the rapid recovery of more opportunistic species. Community differences can also be attributed to low-flow years where species may be highly competitive (fig. 57). Community metrics were compared temporally and spatially—(1) between spring and fall index periods, (2) among years of significant flow, (3) among the Verde River and its tributaries, and (4) among areas with and without significant diversions (table 12). Richness and diversity measures are significantly different between spring and fall index periods, especially in years where high flows have occurred (fig. 57). The purpose of such analyses is to better understand major hydrologic differences before relating community metrics to flow metrics. About 12 macroinvertebrate metrics were found to be significantly different between one or more of the variables tested (index period, high-flow or low-flow sample, diversions present, or if the sample is from the Verde River or a tributary; table 12).

Table 12. Fish microhabitat categories for water velocity, water depth, and substrate, as well as subcategory ranges, descriptors, and quantiles for sample reaches in the middle Verde River and two tributary streams, Arizona.

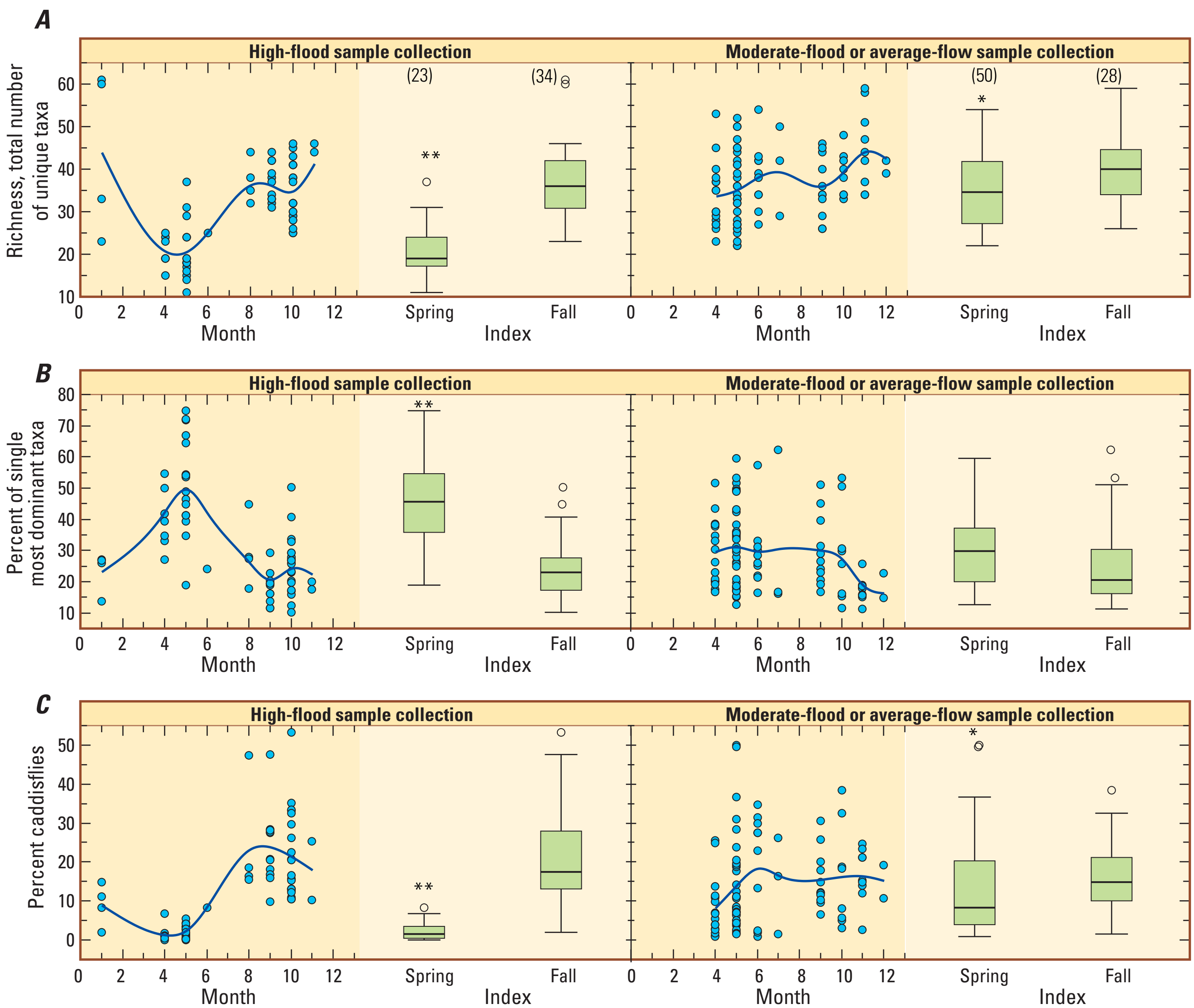
[Table 12 is available online only as a comma separated value (.csv) file and an Excel (.xlsx) file at https://doi.org/10.3133/sir20175100]

The variables providing the greatest differentiation among community metrics were index period, high-flow year, and collection location (Verde River or tributary). Most of the effects observed were with richness (total number of taxa) and diversity measures (for example, Shannon's diversity index or percentage of abundance comprised of the most dominant taxa). Although richness, caddisfly (order Trichoptera) richness, Shannon's diversity index, and the percent of the most dominant taxa are all highly correlated, their response to each variable was different. The percentage of mayfly abundance (order Ephemeroptera) was significantly higher in the upper-middle and lower-middle reaches of the Verde River and in Oak Creek, which also have higher flows and more urbanization than other reaches. The other tributary reaches tended to have higher richness-metric measures and greater caddisfly and midge (order Diptera) taxa richness. An earlier study at Oak Creek demonstrated that spring discharge plays a selective role in larval success and diversity in caddisflies (Blinn and Ruiter, 2009) and supports the idea that long-term monitoring should rely on a single, consistent index period. Many of the variables analyzed are related to each other as well as to streamflow and should be considered in future studies.

Watershed Macroinvertebrate Distribution Using a Reach Characterization with Multiagency Data

Macroinvertebrate samples were collected from the upper-middle and lower-middle reaches of the Verde River and from the major Verde River tributaries of Oak Creek, Wet Beaver Creek, and West Clear Creek. The samples were compiled and analyzed for each reach. Four relative abundance (percent) metrics and five feeding guilds were selected to characterize the macroinvertebrate community of each reach. The orders Ephemeroptera, Trichoptera, and Diptera represent mayfly, caddisfly, and true fly taxa, respectively. Non-insect macroinvertebrate taxa consist of snails, clams, scuds, crayfish, water mites, and aquatic worms. These community metrics are often used to assess sensitivity to or tolerance of water quality condition. Likewise, the feeding guild metrics can provide information on functional composition of the community as macroinvertebrates respond to factors such as nutrient loading and sedimentation. Water quality is partially a function of flow and partially a function of habitat conditions making flow characteristics an important factor affecting community structure. The metrics selected effectively are the response to a myriad of physical and chemical conditions.

General patterns among the Verde River reaches and the major tributaries of Oak Creek, Wet Beaver Creek, and West Clear Creek (fig. 58*A*) showed that median relative abundance of mayfly taxa (order Ephemeroptera) is 36 to 48 percent of the overall composition of all macroinvertebrates collected in the upper-middle and lower-middle Verde River and Oak Creek. The percentage of taxa that are true flies (order Diptera) was higher in the tributaries than in the Verde River reaches and comprised a third or more of the median relative abundance at each. Caddisfly (order Trichoptera) and non-insect taxa had similar percentages among all the reaches suggesting that mayfly and true fly taxa are more influential to function of macroinvertebrate community in the middle Verde River watershed. Macroinvertebrates were also analyzed as functional feeding guilds. Trophic composition and nutrient cycling play a major role in stream ecosystem processes. The feeding guilds were compared at the same Verde River reaches and tributary sites (fig. 58*B*). Collector-gatherers were the most abundant feeding group found at all the reaches followed



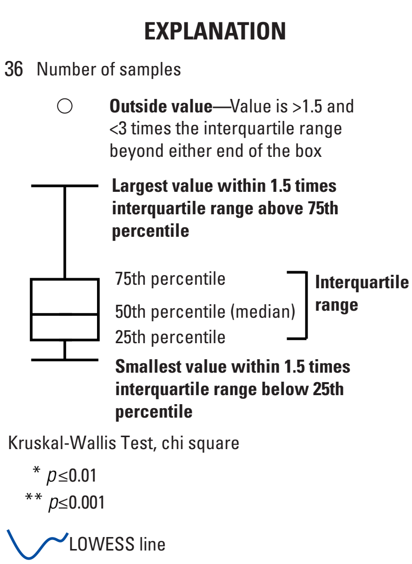


Figure 57. Graphs showing three macroinvertebrate metrics split into flood-magnitude category and plotted by month (number) and index (spring and fall) for the middle Verde River and two tributary streams, Arizona. *A*, Total richness (number of unique taxa); *B*, percentage of abundance represented by the single, most dominant taxa; and *C*, percent of individuals that are caddisflies (order Trichoptera). See figure 27 for graphs of mean daily stream discharge rates.

by the filterers-collectors. Although the composition of the feeding guilds was mostly similar among reaches, the tributaries showed greater percentages of scrapers and shredders. This is because the smaller tributary streams have a greater percent area with shallow substrates than does the Verde River. As a result, more light reaches the bed of the shallow streams, which leads to greater algae growth for consumption by the scrapers. There is also a greater concentration of canopy detritus along the margins of the smaller streams, and this provides more food material for the shredders. Overall, the community metrics did not indicate major differences that could be attributed to differences in flow conditions.

Macroinvertebrate assemblages were investigated because the community metrics were mostly similar among the primary reaches. Additional tributary reaches that had fewer samples were added to the assemblage analysis to determine how similar the species are at each reach. The NMDS ordination of assemblages showed separation between the tributaries and Verde River reaches and similarities in assemblage structure among most of the tributaries except for Bitter Creek, which plotted furthest from the other study reaches (fig. 59). The West Clear Creek, Oak Creek, and Wet Beaver Creek tributaries all have similar relative abundances of each of the taxa groups presented. The Verde River reaches were differentiated by greater abundances of different mayfly taxa, whereas Bitter Creek showed higher abundances of different true fly (order Diptera), midge (family Chironomidae), and noninsect taxa. Bitter Creek is affected by wastewater contributions, and the creek has low dissolved oxygen and high nutrients, which are likely the cause for these more environmentally tolerant groups comprising more than half of the taxa composition (fig. 59).

Associations Among Flow Metrics and Macroinvertebrate Metrics

Sampling locations where streamflow data were available to compute flow metrics were mainly located along the major tributaries of the Verde River—Oak Creek, Wet Beaver Creek, and West Clear Creek. Flow metrics were computed for 30, 90, and 365 days before the collection of macroinvertebrate samples. Correlation scatter plots were used to identify significant relations using a Kendall's tau coefficient ($p \leq 0.05$). Significant relations were associated between macroinvertebrates and the 30-day flow metrics, unlike the fish which were mostly correlated with the longer period flow metrics. This time-correlation difference between fish and macroinvertebrates is likely related to the rapid colonization and lifecycle of macroinvertebrates (Konrad and others, 2008b; Wiele and others, 2012). Correlation analysis of 30-day flow metrics with macroinvertebrate metrics identified several flow metrics related to magnitude and rate-of-change that may influence community structure. Correlated flow-metric examples include the coefficient of variation of daily flows, median maximum annual streamflow, and normalized range of monthly streamflow.

Oak Creek had the largest number of significant associations between macroinvertebrate and flow metrics (fig. 60; table 13). Among the correlations, total richness and caddisfly richness had a negative association to maximum monthly streamflow, normalized range of monthly streamflow, and the 50th percentile of absolute value of daily change in streamflow, which suggests some limiting threshold for caddisfly diversity and relative abundance for certain magnitude flows and rate-of-change in streamflow. The increase in the overall richness was positively associated with the base-flow recession coefficient. The flow and macroinvertebrate relations are confounded by the timing of index sampling period and richness—the fall subset of samples has lower richness than the spring samples.

Wet Beaver Creek samples had a similar pattern of caddisfly richness as Oak Creek but showed increasing associations between total richness and mayfly relative abundance. Functional feeding-group relations were also significant at Wet Beaver Creek, but this may be related to trophic patterns related to season rather than flow as shown by the separation (along the y-axis) between fall and spring samples (fig. 60). The percentage of collector-gatherers relative to overall taxa richness was greater in the fall, and the percentage of scrapers relative to overall taxa richness was greater in the spring. This seasonal difference was also evident in the general plot of percentage of individuals that comprise the caddisfly order compared by index period (fig. 57).

West Clear Creek had the least number of significant relations among macroinvertebrate and hydrologic metrics. Caddisfly relative abundance and taxa showed significant relations to multiple discharge exceedance thresholds (25th, 50th, 75th, 90th, and 99th percent), which suggests a general overall decrease of caddisfly abundance to increased flow and no threshold to a specific magnitude of flow.

Macroinvertebrate Quantile Regression

The quantile-regression analysis helped to show the response of macroinvertebrate metrics in relation to flow metrics that was not apparent using least squares regression. The relative abundance and richness measures of most of the taxa and community metrics other than caddisflies and mayflies were not significantly related to most of the hydrologic metrics analyzed. Several of the hydrologic metrics developed to measure flow variability were correlated, so only the log-transformed median annual maximum flows (30 days before sampling) were evaluated using quantile regression. The median annual maximum flows (30 days before and after log transformation) are shown in relation to the relative abundance of caddisflies and mayflies for all gaged locations (fig. 61*A*, *B*). Although caddisflies and mayflies showed significant responses to hydrologic metrics, they are inversely related ($\tau = -0.38$, $p = 0.0009$) and as a result, the response to the hydrologic metrics was also inversely related. The relative abundance of caddisflies decreased in response to the hydrologic metrics representing streamflow variability, including median annual

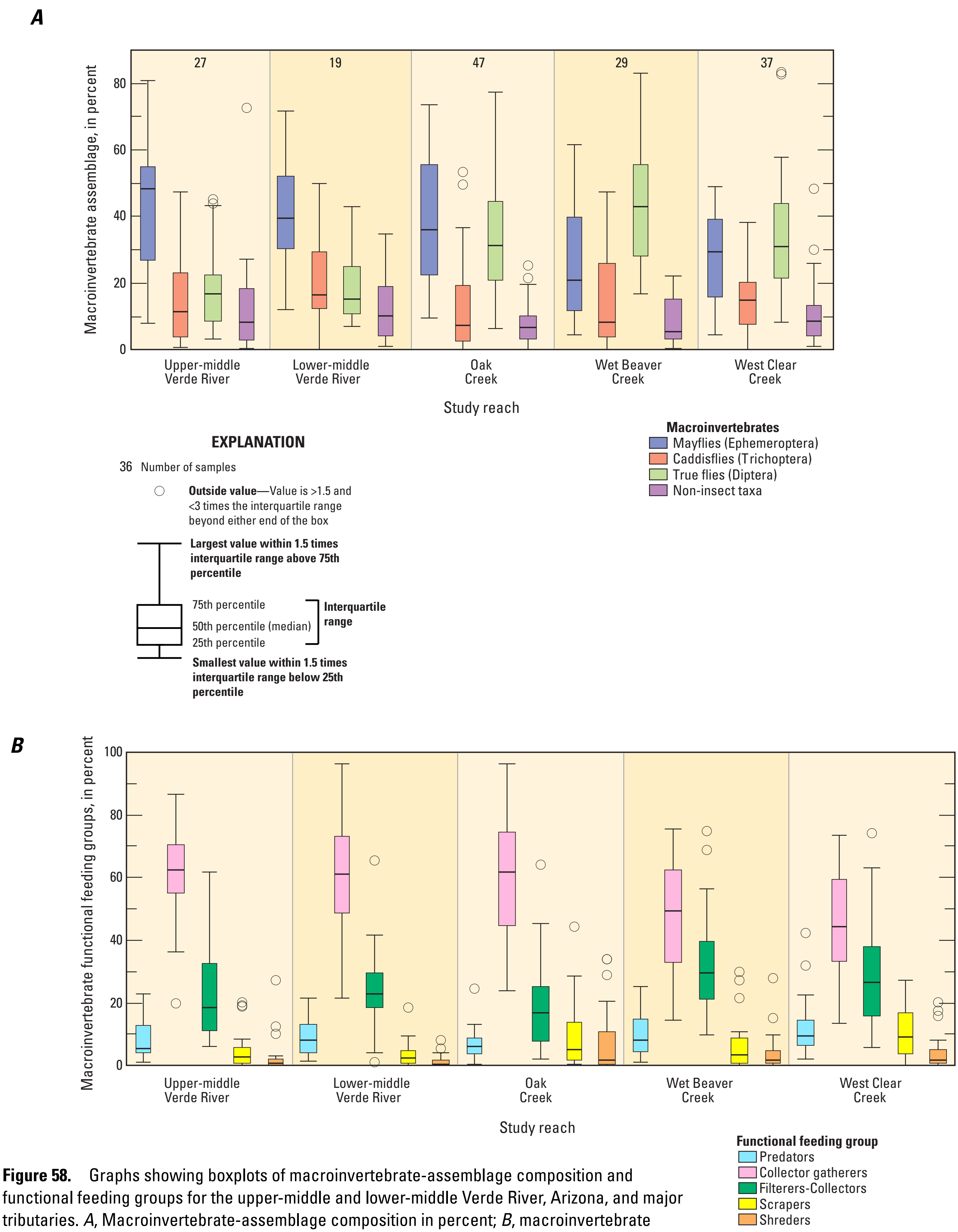


Figure 58. Graphs showing boxplots of macroinvertebrate-assemblage composition and functional feeding groups for the upper-middle and lower-middle Verde River, Arizona, and major tributaries. *A*, Macroinvertebrate-assemblage composition in percent; *B*, macroinvertebrate functional feeding groups in percent.

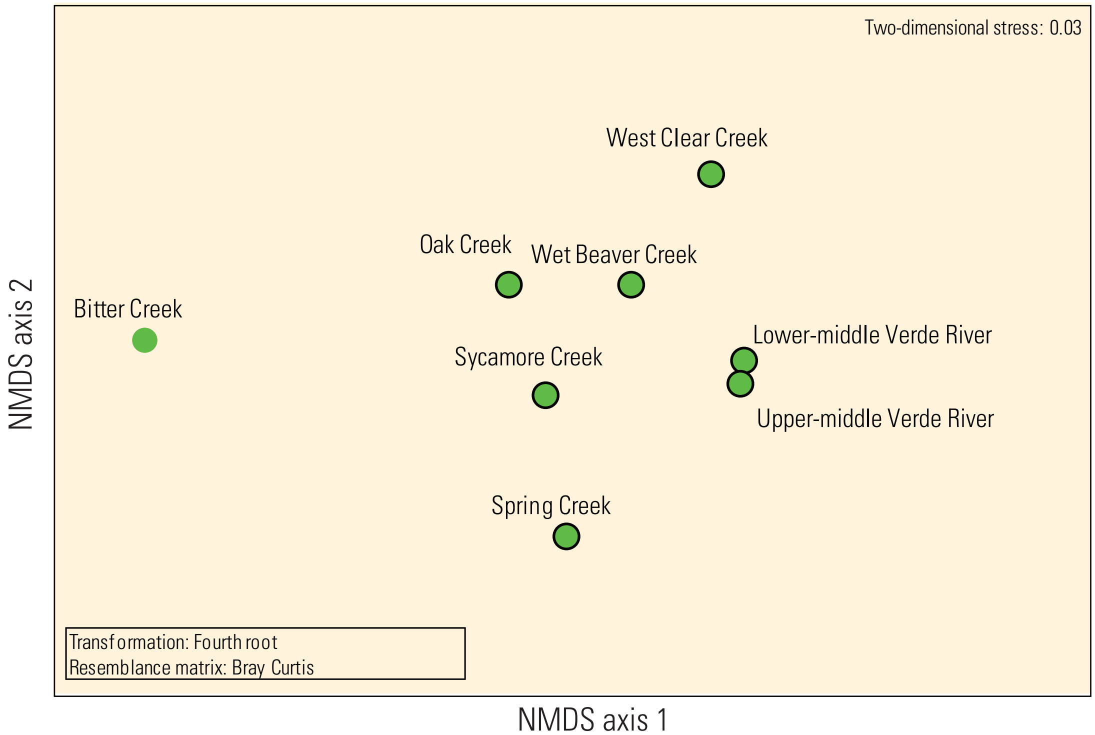


Figure 59. Plot showing nonmetric multidimensional scaling (NMDS) of multivarite centroids for macroinvertebrate assemblages from the upper-middle and lower-middle Verde River, Arizona, and tributaries.

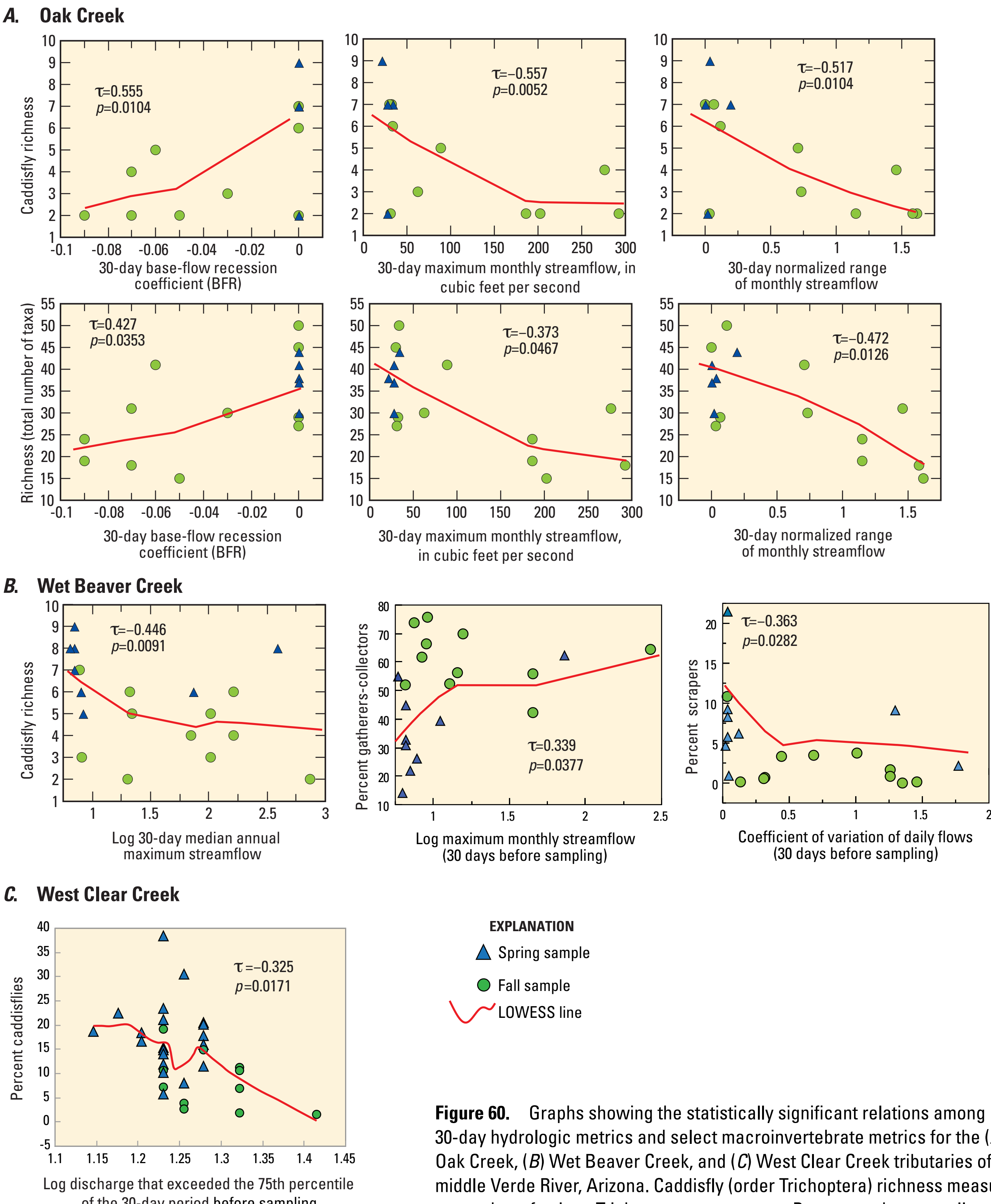


Figure 60. Graphs showing the statistically significant relations among 30-day hydrologic metrics and select macroinvertebrate metrics for the (*A*) Oak Creek, (*B*) Wet Beaver Creek, and (*C*) West Clear Creek tributaries of the middle Verde River, Arizona. Caddisfly (order Trichoptera) richness measured as number of unique Trichoptera taxa present. Percent gatherers-collectors and percent scrappers—percent of overall taxa in the functional feeding group. Percent caddisflies—percent of overall taxa that are caddisflies. τ, Kendall's tau correlation.

Table 13. Macroinvertebrate metrics significantly greater ($p \leq 0.05$) in reaches of the middle Verde River and two tributaries, Arizona, that have water diversions and were collected during the spring or fall index period, collected after large magnitude floods or no large flooding events, and collected on the Verde River or a tributary.

[--, no significant difference]

Macroinvertebrate metric	Stream diversion	Index period	Large-flood year	Mainstem or tributary
Total number of taxa (richness)	--	Greater in the fall	Greater after no large-magnitude flooding	Greater in the samples collected in tributaries
Total mayfly (Ephemeroptera) taxa	--	--	--	Greater in the samples collected in the mainstem
Total caddisfly (Trichoptera) taxa	--	Greater in the fall	--	--
Percent of abundance composed of mayflies	--	--	Greater after large-magnitude flooding	Greater in the samples collected in the mainstem
Percent of abundance composed of caddisflies	--	Greater in the fall	--	--
Percent of abundance composed of true flies (Diptera)	--	Greater in the spring	--	Greater in the samples collected in tributaries
Percent of abundance composed of non-biting midges (Orthocladiinae)	--	--	Greater after no large-magnitude flooding	Greater in the samples collected in tributaries
Percent of abundance composed by non-insects	Greater in diverted reaches	--	Greater after no large-magnitude flooding	--
Percent of abundance composed by worms in the subclass Oligochaetes	Greater in diverted reaches	--	--	--
Shannon diversity index	--	Greater in the fall	Greater after no large-magnitude flooding	--
Total abundance	--	Greater in the spring	--	Greater in the samples collected in the mainstem
Percent of abundance composed by pollution tolerant taxa	--	--	Greater after no large-magnitude flooding	Greater in the samples collected in tributaries

maximum flows 30-days before sampling (fig. 61*A*), while relative abundance of mayfly increased (fig. 61*B*).

A majority of the caddisfly relative-abundance density decreased (quantiles 0.25–0.60) with increasing median annual maximum flow values. A proportion of the density (>0.75) is less responsive the higher median annual maximum flow values, possibly indicating certain species of caddisflies are less affected by the larger median annual flows (fig. 61*A*). The mayflies relative abundance showed that about half of the density (quantiles 0.50–0.90) responded similarly, increasing with increased median annual maximum flow values 30-days before sampling , indicating preference for flow variability (fig. 61*B*).

Caddisfly population relative to median annual maximum flows 30-days before sampling showed no floor and a poorly defined ceiling, following more of the classic wedge-shaped scatterplot distribution (Konrad and others, 2008b; Wang and others, 2007; Kail and others, 2011). In other words, caddisflies were more common in lower flows but did not tolerate the highest flows (fig. 61*A*). This implies that at lower median annual maximum flows other factors predominate in limiting overall caddisfly abundance. At higher flows, median annual maximum flow plays a greater role in limiting their relative population.

Ceilings and floors (Konrad and others, 2008b) were both evident in the mayfly data (fig. 61*B*). The ceiling (decreasing 0.90 quantile) implies that additional factors other than antecedent streamflow conditions alone limit the presence of these macroinvertebrates at low median annual maximum flow values. In low flows (<1.0 median annual maximum flows, 30 days before sampling), no more than 50 to 60 percent of total macroinvertebrates per sample were mayflies. On the other hand, higher median maximum annual flow values may be more of a limiting factor for macroinvertebrate species other than mayflies, which could allow mayflies to outcompete these species. This would explain the increasing percent of mayflies with median maximum annual flow , 30 days before sampling in even the lowest quantile (floor) (fig. 61*B*).

Caddisfly populations appeared to be responding to less flashy and more stable streamflows, whereas the opposite was true for mayfly populations. The results of the quantile regression also suggest that estimating effects of hydrologic metrics based on only the mean may underestimate or overestimate the effects observed by quantiles above the mean.

Macroinvertebrate Pilot Study

In 2010, six macroinvertebrate samples were collected from the Verde River in the upper part of its middle watershed using USGS protocols (Moulton and others, 2002). The six locations were the Verde River below Granite Creek, Salt River Project low-flow streamgage, USGS Verde River at Paulden gage, Perkinsville Bridge, USGS Verde River at Clarkdale gage, and Reitz Ranch property (fig. 26). The NMDS ordination of macroinvertebrate assemblages showed that the Verde River below Granite Creek site was different than all the other sites sampled and that Perkinsville and Reitz Ranch samples had similar community structure (fig. 62). The Verde below Granite Creek had the largest percentage of caddisfly taxa, lowest mayfly taxa, and lowest diversity (Shannon diversity index of 0.91; largest Shannon diversity index of 1.29 was at Clarkdale) of all the locations. Several taxa were observed at the Verde River below Granite Creek that were not found at any of the other five locations (table 14). Paulden had high mayfly taxa percentages but very low midge taxa percentages, whereas the other four stations were similar in overall diversity and caddisfly taxa percentages.

Table 14. Kendall tau correlations between macroinvertebrate metrics and hydrologic metrics for the middle Verde River, Arizona, and tributaries.

[Table 14 is available online only as a comma separated value (.csv) file and an Excel (.xlsx) file at https://doi.org/10.3133/sir20175100.]

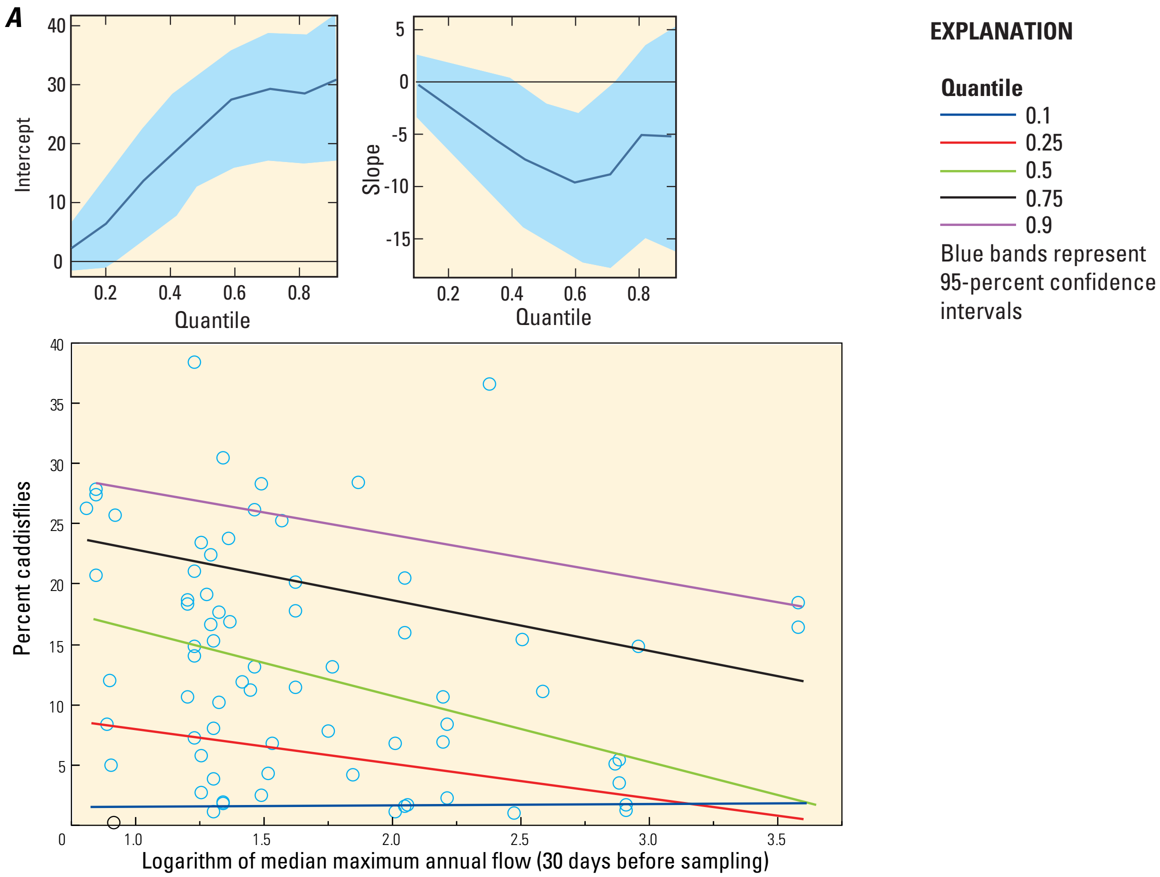
Superimposing microhabitat variables on the NMDS ordination revealed differences between sites and the communities found there (fig. 62). The differences in community composition at the Verde River below Granite Creek location compared to the other sites are likely due to the smaller substrate sizes, lower velocities, shallow depths, and highest riparian cover. The Campbell Ranch and Paulden locations had similar microhabitat characteristics, with the exception of riparian cover, yet the assemblage structure was very different.

Conclusion and Future Directions

The middle Verde River watershed is complex in ecological function exhibiting characteristics of an undisturbed flow regime in the upper part and an altered flow regime in the lower part, a result of surface-water diversions and groundwater withdrawal. Relatively unaltered flow in the upper-middle watershed maintains a healthy ecosystem, with a mixed age, mixed patch vegetative structure. In the river, more riffle habitat and fewer diversions were present in the upper-middle part of the watershed, whereas more pools and diversions were present in the lower-middle part. In heavily diverted reaches, non-native fish outnumbered native fish, whereas more native fish species were found in the nondiverted reaches. Generally, the greatest number and relative abundance (percentage) of native fish were also found in the upper-middle reaches, and non-native fish generally declined in presence with distance downstream. Consistent with these observations is that across five locations sampled in the middle Verde watershed, native fish preferred slow to moderate velocity moderately deep water over coarse gravel, whereas non-native fish preferred standing, deeper water over silt and sand, although non-native species were also found in faster, shallow water over coarse gravel. Native suckers (desert and Sonora) in particular decline during periods of extended base flow, whereas non-native sunfish (bass species and green sunfish) in contrast decline during periods of flashy, high flows. Among macroinvertebrates, caddisfly populations appeared to prefer less flashy, more stable streamflows, whereas mayfly populations responded to less stable, more flashy streams. Overall, it appears that unaltered disturbance regimes with fewer diversions and more flashy and somewhat higher velocity flows are generally preferred by native species in the middle Verde River watershed.

The statistically based empirical relations determined in this report can be used to develop the basis for a conceptual model of environmental flows in the middle Verde River watershed. A robust conceptual model can provide the foundation for developing physically based, quantitative and predictive models of environmental flows for this semiarid southwestern river. Such models can improve the understanding of the effects of increased stresses from climate variability and increased demands on the water supply of the Verde River.

Spring snowpack across the western United States (including the Southwest) is predicted to decrease sharply through the 21st century, leading to reductions in soil moisture and substantially lower spring-summer runoff and base flow (Cayan and others, 2013). These conditions could result in increased flow intermittency (of both area and frequency), as well as an increase in the total time reaches remain dry. Increased intermittency would affect in-stream biota, including macroinvertebrates (Miller and Brasher, 2011). In addition, a changing climate could affect the magnitude, as well as timing, of all components of the hydrograph (Cayan and others, 2013; Jaeger and others, 2014), further stressing a system that already faces reduction in streamflow as a result of groundwater withdrawals and surface-water diversions. Changes to the flow regime will alter the frequency, timing, and intensity of the mobilization of the bed sediments that form the microhabitat framework. Alterations to hydrologic characteristics have been shown to influence in-stream microhabitat (Brasher and



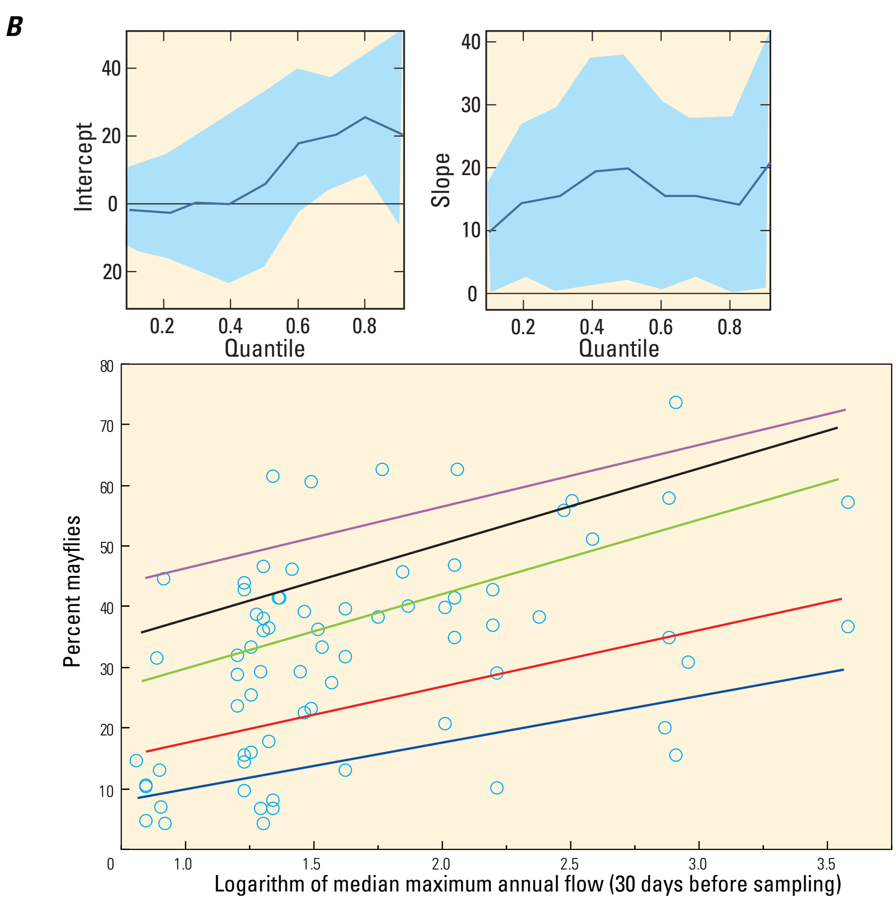


Figure 61. Graphs showing quantile regression lines, intercepts, and slopes of the 0.10, 0.25, 0.50, 0.75, and 0.90 quantile regression lines for significantly correlated hydrologic and macroinvertebrate metrics for the middle Verde River and two tributary sites, Arizona. *A*, Relative abundance of caddisflies (percent of overall taxa that are order Trichoptera) and the log-transformed median annual maximum flow 30 days before sampling. *B*, Relative abundance of mayflies (percent of overall taxa that are order Ephemeroptera) and the log-transformed median annual maximum flow 30 days before sampling. Blue band represent the 95-percent confidence intervals for each plot.

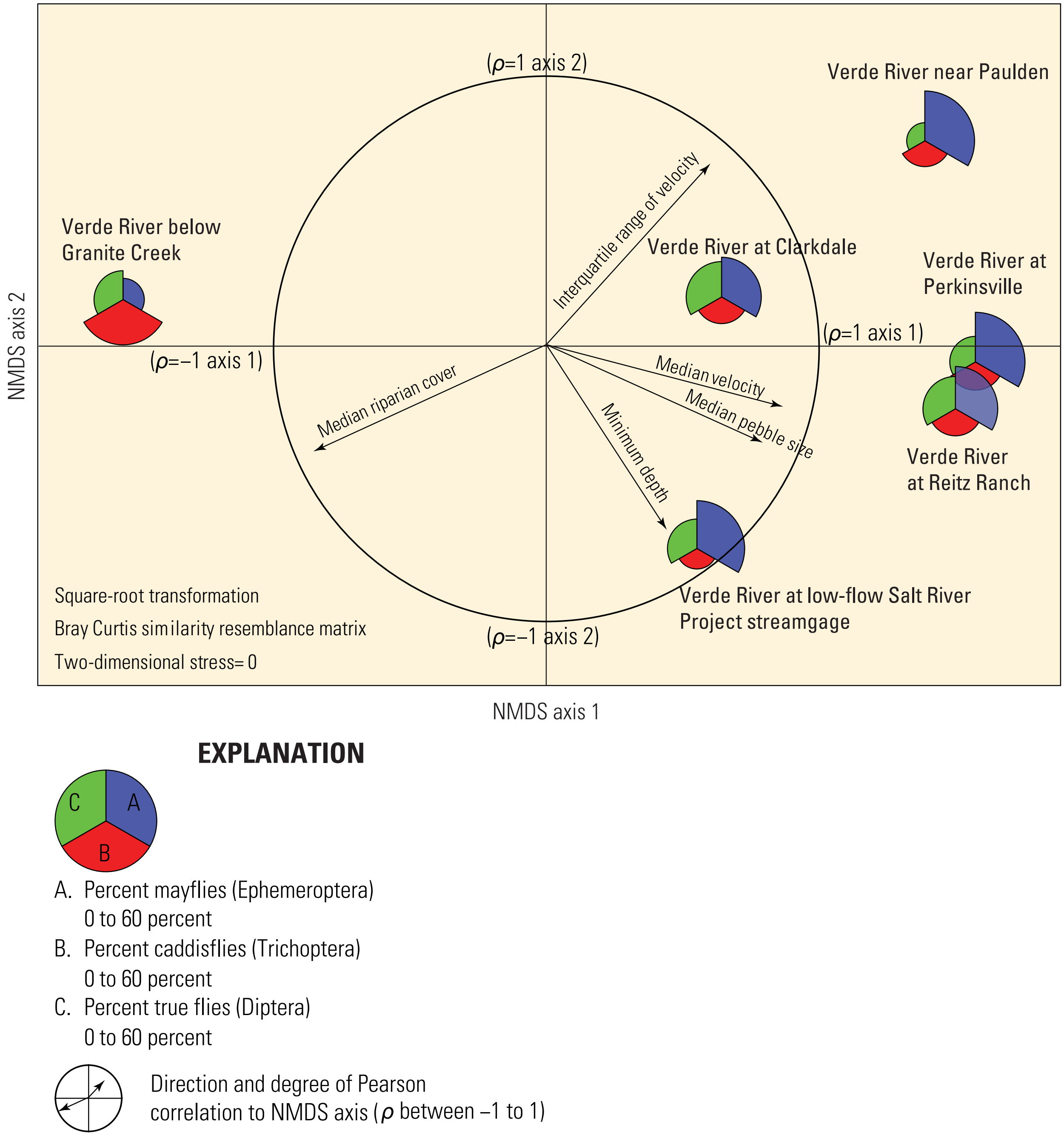


Figure 62. Diagram showing nonmetric multidimensional scaling (NMDS) ordination plot of assemblages and macroinvertebrate metric pie plots of percent mayflies, percent caddisflies, and percent true flies (as a percent of overall taxa) for the middle Verde River, Arizona. Pie plots are located at the respective NMDS assemblage point for Verde below Granite Creek, Salt River Project low-flow streamgage, Paulden streamgage, Perkinsville Bridge, Clarkdale streamgage, and Reitz Ranch property sampling sites on the middle Verde River, Arizona. Size of pie sector corresponds to percent of (A) mayflies, (B) caddisflies, (C) true flies. Vector Spearman correlation lines are plotted (rho, ρ, >0.8) of median riparian cover, minimum depth, median pebble size, median velocity, and interquartile range of velocity.

Table 15. Unique taxa observed in the six macroinvertebrate assemblages collected from the Verde River in the upper part of the middle Verde River watershed, Arizona.

Taxa level identification	Abundance	Site	Taxa order	Common name
Gomphidae	92	Salt River Project low-flow streamgage	Odonata	clubtail dragonflies
Culoptila sp.	4	Salt River Project low-flow streamgage	Trichoptera	caddisfly
Tanytarsus sp.	5	USGS Clarkdale streamgage	Diptera	non-biting midge
Sublettea sp.	27	USGS Clarkdale streamgage	Diptera	non-biting midge
Heptageniidae	11	USGS Clarkdale streamgage	Ephemeroptera	mayfly
Capniidae	19	USGS Clarkdale streamgage	Plecoptera	stonefly, small winter stoneflies
Centroptilum sp.	12	USGS Paulden streamgage	Ephemeroptera	mayfly, small Minnow Mayflies
Ambrysus sp.	48	USGS Paulden streamgage	Hemiptera	creeping water bugs
Hirudinea	2	Verde river below Granite Creek	Annelida	leech
Corynoneura sp.	2	Verde river below Granite Creek	Diptera	European non-biting midges
Paratanytarsus sp.	2	Verde river below Granite Creek	Diptera	European non-biting midges
Stictochironomus sp.	2	Verde river below Granite Creek	Diptera	non-biting midge
Tipula sp.	5	Verde river below Granite Creek	Diptera	crane fly
Cricotopus bicinctus group	10	Verde river below Granite Creek	Diptera	midge
Physa sp.	5	Verde river below Granite Creek	Gastropoda	left-handed or sinistral, air-breathing freshwater snails
Sigara sp.	5	Verde river below Granite Creek	Hemiptera	water boatmen
Ostracoda	14	Verde river below Granite Creek	Ostracoda	seed shrimp
Pisidium sp.	3	Verde river below Granite Creek	Veneroida	minute freshwater clams known as pill clams or pea clams
Microtendipes pedellus group	12	Verde river near Reitz Ranch	Diptera	midge
Eukiefferiella brehmi group	15	Verde river near Reitz Ranch	Diptera	non-biting midge
Pseudorthocladius sp.	15	Verde river near Reitz Ranch	Diptera	European non-biting midges

others, 2004; Konrad and others, 2008; Brasher and others, 2010), and thus fish and macroinvertebrate assemblage structure (Konrad and others, 2008b; Brasher and others, 2010). Additional research can begin to ascertain the hydrological requirements of those plant and animal species that are reliant on the Verde River system.

The primary link between groundwater withdrawals, diversions, and climate change on the one hand and their effects on riparian and in-stream microhabitats on the other is through streamflow. Streamflow alteration caused by human stressors and climate change will manifest in changes to the riparian habitat and streamflow regime. These changes will ultimately lead to shifts in the structure and function of the biotic communities that rely on these habitats. Measuring these changes, attributing causative mechanisms, and developing models to represent future conditions will be challenging. A common approach has been to mine data for correlative associations between biota and streamflow characteristics in an attempt to define the habitat thresholds of species and communities. A more holistic synthesis of the biological and physical components of the middle Verde River watershed (including many of the habitat, biologic, and hydraulic characteristic discussed in this report) would complement the correlative approach and give resource managers a process-based understanding of the links between the geomorphology and microhabitats of the middle Verde River watershed. This would also provide a physical basis for understanding the role of climate change in the development and alteration of riparian and aquatic microhabitat.

Models of flow and sediment transport have been used to examine environmentally damaged and vulnerable rivers (for example, Wiele and others, 2007) and improve management of environmental recovery (for example, Wiele and Torizzo, 2005). Calculations of the hydraulic properties of in-stream microhabitat have been used to examine seasonal variations and effects of dam operation on in-stream microhabitat (Korman and others, 2004). Calculations of the hydraulic geometry and sediment transport at a streamgage site on the upper-middle Verde River have provided insight into the timing and magnitude of the river's response to significant runoff events and the relation between sedimentary processes and in-stream microhabitat. Models can be used to extrapolate information from small-scale hydraulic studies to facilitate a better understanding of relations among low- and medium-scale flows and microhabitat properties, as well as estimate critical flows for reworking of the channel. Recent advances in hydrological field techniques and environmental modeling will also contribute to unraveling these complex linkages (for example, Korman and others, 2004). Research using acoustic Doppler technology (Klaar and others, 2009; Klaar and others, 2011) has provided a means of linking changes in in-stream discharge with the availability of in-stream microhabitat by measuring small-scale near-bed flow fields. These data provide a direct, physical link between discharge and channel properties that was not possible to obtain with cross sectionally or vertically averaged flow velocities or with coarser velocity measurements made with mechanical meters. When these data are combined with microhabitat preferences of fish and macroinvertebrates, it is possible to ascertain how microhabitat availability will change with changing river base flow, and hence identify environmental flow levels and timings that are crucial for species survival.

During this study, pilot studies were conducted to begin to identify associations among hydrologic and biologic metrics. Additional studies would enable the quantification of the response of in-stream and riparian biota to hydrologic conditions and link aquatic-community structure with flow-alteration scenarios. Hydrologic and biologic metrics that are most responsive to flow-alteration could then be identified, and models for flow alteration-ecological response scenarios could be developed.

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Menlo Park Publishing Service Center, California
Manuscript approved for publication September 7, 2017
Edited by James W. Hendley II
Layout and design by Vivian Nguyen

Paretti and others—**Preliminary Synthesis and Assessment of Environmental Flows, Middle Verde River Watershed, Arizona**—Scientific Investigation Report 2017–5100

Desert sucker (*Catostomus clarki*) from West Clear Creek, a tributary to the Verde River

ISSN 2328-0328 (online)
http://dx.doi.org/10.3133/sir20175100