Prepared in cooperation with The Nature Conservancy, the Bureau of Land Management, Cochise County, the City of Sierra Vista, and the U.S. Department of Defense

Hydrological Conditions and Evaluation of Sustainable Groundwater Use in the Sierra Vista Subwatershed, Upper San Pedro Basin, Southeastern Arizona

Scientific Investigations Report 2016–5114
Version 1.2, February 2017

U.S. Department of the Interior
U.S. Geological Survey

COVER. Photograph showing San Pedro Riparian National Conservation Area, residential development southeast of Sierra Vista, Arizona, and the Huachuca Mountains from Hereford Road, Cochise County, Arizona (U.S. Geological Survey photograph by Thomas Porter).

BACK COVER. Photographs of (1) San Pedro River upstream of the Charleston gaging station, Cochise County, Arizona (top left); (2) U.S. Geological Survey scientist making a discharge measurement as part of a water-quality sampling field visit downstream of Murray Springs in Curry Draw, Cochise County, Arizona (bottom left); (3) summer monsoon thunderstorm in Garden Canyon, Huachcua Mountains, Cochise County, Arizona, from behind field of flowers at the Foudy Well, near Palominas, Arizona (top right) (U.S. Geological Survey photographs by Bruce Gungle); and (4) boundary of the San Pedro Riparian National Conservation Area (bottom right) (U.S. Geological Survey photograph by Tom Porter).

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By Bruce Gungle, James B. Callegary, Nicholas V. Paretti, Jeffrey R. Kennedy, Christopher J. Eastoe, Dale S. Turner, Jesse E. Dickinson, Lainie R. Levick, and Zachary P. Sugg

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U.S. Geological Survey

U.S. Department of the Interior
SALLY JEWELL, Secretary

U.S. Geological Survey
Suzette M. Kimball, Director

U.S. Geological Survey, Reston, Virginia
First release: 2016
Revised: October 2016 (ver. 1.1)
Revised: February 2017 (ver. 1.2)

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Suggested citation:
Gungle, Bruce, Callegary, J.B., Paretti, N.V., Kennedy, J.R., Eastoe, C.J., Turner, D.S., Dickinson, J.E., Levick, L.R., and Sugg, Z.P., 2016, Hydrological conditions and evaluation of sustainable groundwater use in the Sierra Vista Subwatershed, Upper San Pedro Basin, southeastern Arizona (ver. 1.2, February 2017): U.S. Geological Survey Scientific Investigations Report 2016–5114, 90 p., https://doi.org/10.3133/sir20165114.

ISSN 2328-0328 (online)

Acknowledgments

The authors are grateful to the Upper San Pedro Partnership for the long-term support of the Sierra Vista Subwatershed monitoring program. The U.S. Geological Survey (USGS) initiated this report, but it could not have been completed without the additional generous support of The Nature Conservancy, Bureau of Land Management, Cochise County, the City of Sierra Vista, and the U.S. Army's Fort Huachuca. In addition, the Agricultural Research Service provided in-kind support for parts of the monitoring program on which this report relies.

The report benefited greatly from in-depth technical discussions with members of the Upper San Pedro Partnership Technical Committee, including Dr. Russell Scott and Dr. David Goodrich (U.S. Department of Agriculture, Agricultural Research Service), Brooke Bushman (The Nature Conservancy), Scott Dooley (City of Sierra Vista), Tricia Gerrodette (Arizona Audubon), Ben Lomeli (Bureau of Land Management), Karen Modesto (Arizona Department of Water Resources), Kim Mulhern (Cochise County), and Justin Nixon (Fort Huachuca). Julian Heilman and Amy Read of the USGS were extremely helpful in creating a number of the figures. Gene Fenstermacher (City of Sierra Vista Environmental Affairs Commission) contributed greatly with a number of probing questions, and Linda Stitzer (Western Resource Advocates) shared valuable and hard-to-find data, as well as invaluable institutional knowledge.

Don Pool (USGS) has an encyclopedic knowledge and detailed understanding of the Upper San Pedro Basin that he willingly shared in numerous discussions. Stan Leake (USGS) patiently explained a number of complex hydrology and modeling concepts. Holly Richter (The Nature Conservancy) has been a tireless advocate for unbiased science and hydrologic monitoring in the Upper San Pedro Basin, and to her we owe a great debt for the scientific progress that has been made there. Finally, thanks to Andre Newcomb for his abiding interest in hydrologic issues across the Sierra Vista Subwatershed.

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Conversion Factors

Inch/Pound to International System of Units

Multiply	By	To obtain
	Length	
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 mile (mi^2)	259.0	hectare (ha)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
gallon (gal)	3.785	liter (L)
gallon (gal)	0.003785	cubic meter (m^3)
cubic foot (ft^3)	0.02832	cubic meter (m^3)
acre-foot (acre-ft)	1,233	cubic meter (m^3)
acre-foot (acre-ft)	0.001233	cubic hectometer (hm^3)
	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 year (ft/yr)	0.3048	meter per year (m/yr)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
gallon per day (gal/d)	0.0224	acre-foot per year (acre-ft/yr)
gallon per day (gal/d)	0.003785	cubic meter per day (m^3/d)
million gallons per day (Mgal/d)	0.04381	cubic meter per second (m^3/s)
	Mass	
ton per year (ton/yr)	0.9072	metric ton per year
	Hydraulic conductivity	
foot per day (ft/d)	0.3048	meter per day (m/d)
foot per day (ft/d)	111.3282	meter per year (m/yr)

Multiply	By	To obtain
	Acceleration	
microGal (μGal)	10	nanometer/second^2 (nm/s^2)
microGal (μGal)	0.328×10^{-9}	feet/second^2 (ft/s^2)

International System of Units to Inch/Pound

Multiply	By	To obtain
	Length	
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
hectare (ha)	0.003861	square mile (mi^2)
	Volume	
liter (L)	0.2642	gallon (gal)
cubic meter (m^3)	0.0008107	acre-foot (acre-ft)
cubic hectometer (hm^3)	810.7	acre-foot (acre-ft)
	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)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
cubic meter per day per square kilometer [(m^3/d)/km^2]	0.0006844	million gallons per day per square mile [(Mgal/d)/mi^2]
	Mass	
metric ton per year (t/yr)	1.102	ton per year (ton/yr)
	Hydraulic conductivity	
kilograms per hectare per year [(kg/ha)/yr]	0.8921	pounds per acre per year [(lb/acre)/yr]

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as °F = (1.8 × °C) + 32.

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as °C = (°F – 32) / 1.8.

Datum

Vertical coordinate information is referenced to the [insert datum name (and abbreviation) here, for example, North American Vertical Datum of 1988 (NAVD 88)].

Horizontal coordinate information is referenced to the [insert datum name (and abbreviation) here for example, North American Datum of 1983 (NAD 83)].

Altitude (or elevation), as used in this report, refers to distance above the vertical datum.

Supplemental Information

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (μS/cm at 25°C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

Concentrations of chemical constituents in water are estimated and given in nanogram of compound per liter of water (ng/L). Compound concentrations measured in extracts are reported in nanogram per ampoule of extract from a composite of three semipermeable membrane devices (SPMD) or three polar organic chemical integrative sampler (POCIS) media in each sampler.

Change in the acceleration due to gravity is given in µGal per day (1×10^{-8} m/s^2/day)

Note to U.S. Geological Survey Users: Use of hectare (ha) as an alternative name for square hectometer (hm^2) is restricted to the measurement of small land or water areas. Use of liter (L) as a special name for cubic decimeter (dm^3) is restricted to the measurement of liquids and gases. No prefix other than milli should be used with liter. Metric ton (t) as a name for megagram (Mg) should be restricted to commercial usage, and no prefixes should be used with it.

Abbreviations

ADEQ	Arizona Department of Environmental Quality
ADWR	Arizona Department of Water Resources
ANOVA	analysis of variance
ARS	Agricultural Research Service
BLM	Bureau of Land Management
EGRET-WRTDS	Exploration and Graphics for River Trends
EMAP	Environmental Monitoring and Assessment Program
EOP	City of Sierra Vista's Environmental Operations Park
EPA	U.S. Environmental Protection Agency
ET	evapotranspiration
GMWL	global meteoric water line
gpcd	gallons per capita per day
NAWQA	USGS National Water-Quality Assessment Program
NWIS	USGS National Water Information System
Partnership	Upper San Pedro Partnership
per mil	one part in one thousand parts or 1/1,000
POCIS	polar organic chemical integrative samplers
SPMD	semipermeable membrane devices
SPRNCA	San Pedro Riparian National Conservation Area
TOC	trace organic compound
USGS	U.S. Geological Survey

Hydrological Conditions and Evaluation of Sustainable Groundwater Use in the Sierra Vista Subwatershed, Upper San Pedro Basin, Southeastern Arizona

By Bruce Gungle, James B. Callegary, Nicholas V. Paretti, Jeffrey R. Kennedy, Christopher J. Eastoe,[1] Dale S. Turner,[2] Jesse E. Dickinson, Lainie R. Levick,[3] and Zachary P. Sugg[1]

Abstract

This study assessed progress toward achieving sustainable groundwater use in the Sierra Vista Subwatershed of the Upper San Pedro Basin, Arizona, through evaluation of 14 indicators of sustainable use. Sustainable use of groundwater in the Sierra Vista Subwatershed requires, at a minimum, a stable rate of groundwater discharge to, and thus base flow in, the San Pedro River. Many of the 14 indicators are therefore related to long-term or short-term effects on base flow and provide us with a means to evaluate groundwater discharge to and base flow in the San Pedro River. The indicators were based primarily on 10 to 20 years of data monitoring in the subwatershed, ending in 2012, and included subwatershedwide indicators, riparian-system indicators, San Pedro River indicators, and springs indicators.

Groundwater management actions including voluntary retirement of irrigation pumping in the subwatershed resulted in about a 5,100 acre-feet (acre-ft) reduction in net human use from 2002 to 2012. Subwatershed population increased more than 10,000 during the same period. Most of the reduction occurred during 2002–07 and included reductions in groundwater pumping and increases in managed recharge; net human use varied annually by a few hundred acre-ft during 2007–12. The groundwater budget for 2012 showed a deficit of about 5,000 acre-ft, although the total water-budget uncertainty was about 5,500 acre-ft.

In the vicinity of the U.S. Army's Fort Huachuca, regional-aquifer water levels were in steady decline beginning in at least the mid-1990s (in older wells since at least the early-1970s), as the cone of depression centered on the Sierra Vista and Fort Huachuca pumping centers continued to deepen. This was evident in the individual water levels on Fort Huachuca, as well as from the horizontal hydraulic gradients that extend from the pumping centers toward the San Pedro and Babocomari Rivers. Basin water levels in wells southeast of Sierra Vista, away from the river, were also experiencing declines, while some water levels closer to the river were rising.

Near-stream vertical gradients along the San Pedro River showed no clear increasing or decreasing trends that would indicate a shift in the direction of subsurface flow between the riverbed and the alluvial aquifer, or a trend in the magnitude of groundwater/surface-water exchange. Annual streamflow permanence data showed no clear change in streamflow permanence trends in any of the river reaches, other than those related to precipitation trends. Similarly, the single-day, dry-season, wet-dry streamflow analysis of all subwatershed river reaches indicated no change in condition over the past 14 years, with the exception of the Hereford reach, which has seen a statistically significant increase in wetted length. Dry-season, alluvial-aquifer water levels in the Hereford reach also showed a statistically significant increase. These improvements are attributed to the end of irrigation pumping in the area. Although data indicate that the length of the Fairbank North wetted reach may be in decline, it is not yet statistically significant.

Stable-isotope data indicated reduced groundwater discharge to the Babocomari River in the vicinity of the Babocomari River near Tombstone gaging station and to the San Pedro River near the San Pedro River at Palominas gaging station and near the Lewis Springs DCP stage recorder. The Babocomari River near Tombstone gaging station is downgradient of the major pumping centers. The change in isotopic signature at the Lewis Springs stage recorder could have been the result of alterations in groundwater/surface-water interactions there caused by beaver damming of the river. Base flow in the San Pedro River declined over the periods of record at the three San Pedro River gaging stations in the subwatershed (Palominas, Charleston, and Tombstone), as well as at the Babocomari River near Tombstone gaging station. Precipitation declined slightly from the 1990s to the 2000s, although there is no statistically significant trend in subwatershed precipitation from 1991 to 2012. The occurrence of large winter discharge events appeared to decline and that of large summer discharge events appeared to increase over this same period.

Data for physical parameters, general chemistry, nutrient species, select trace elements, and suspended sediment were collected at San Pedro River at Charleston stream-gaging station. These data were summarized over time and analyzed in relation to discharge and season as a means to assess trends over the period of analysis. Federal and State of Arizona drinking-water and human-contact standards were all met and few exceedances occurred for the ecological thresholds investigated. Several constituents showed a significant trend over the period of analysis, but only concentration and flux data for total phosphate, orthophosphate,

[1]University of Arizona.

[2]The Nature Conservancy.

[3]U.S. Department of Agriculture.

total nitrogen, suspended sediment, and sulfate were suitable to be used in a weighted regression analysis that statistically accounted for time, discharge, and season. Sulfate concentrations and flux showed a significant downward trend over the period of analysis, whereas total phosphorus and ortho-phosphate showed a relatively small magnitude upward trend relative to standards. Suspended sediment concentrations and flux both showed a significant downward trend in the 1980s, an effect attributed to reduction of cattle in the subwatershed at about this time, and (or) increased cottonwood (*Populus fremontii*) and willow (*Salix goodingii*) recruitment, and (or) the curtailment of sand and gravel mining adjacent to the San Pedro River with the designation of the San Pedro Riparian National Conservation Area in 1988. A spike in sediment flux in 2006 may be attributable to the more than 100 debris flows in the Huachuca Mountains during the summer monsoon of that year.

Spring discharge along the San Pedro River generally increased at three sites proximate to the Sierra Vista treated effluent recharge facility and varied somewhat with climate at two other sites. Median annual discharge at the recharge facility peaked in 2006, and at Murray Springs and Horsethief Spring, downgradient of the recharge facility, in 2009. Sampling for trace organic compounds in flow from springs was carried out using both discrete sampling and passive sampling methods. Spring samples thus collected showed the presence of trace-organic compounds. Lewis Springs (background site) had the least number of detections, whereas Murray Springs, located directly downgradient of the City of Sierra Vista's treated effluent recharge facility, had the greatest number of detections of all the springs. Discrete samples from the recharge facility had more than twice the detections found in discrete samples from Murray Spring and at much higher concentrations. Few similar trace-organic compounds were detected at both the springs and the treated effluent recharge facility, and the number of detections did not increase during the collection period. Limitations of the study prevented the determination of trace-organic concentration in passive samplers and also prevented linking trace organic compounds detected at the treated effluent recharge facility with compounds detected from the springs. In particular, trace organic compounds could also derive from other sources such as septic systems.

Looking at the subwatershed as a whole, base flow was in decline along the entire river reach, but determination of the specific cause of the decline was beyond the scope of this report. Conditions in the area from the municipal pumping center of Sierra Vista and Fort Huachuca northeast to the river (from about the Charleston to Tombstone gaging stations) were more commonly in decline than in regions further south. Both long-term indicators, such as regional aquifer groundwater levels and horizontal gradients, and the isotope analysis indicated that groundwater discharge to the river and thus base flow may continue to decline in that area. South of Charleston, indicators were more mixed. Some indicators in the Hereford reach suggest groundwater discharge to the San Pedro River may be increasing there, whereas some indicators in the Palominas reach suggest groundwater discharge to the river there may be declining.

Introduction

In the introduction to the U.S. Geological Survey Circular "Sustainability of ground-water resources," Alley and others (1999) presented a definition of groundwater sustainability that is anchored to three components or pillars of sustainability:

> . . . we define groundwater sustainability as development and use of ground water in a manner that can be maintained for an indefinite time without causing unacceptable environmental, economic, or social consequences. The definition of 'unacceptable consequences' is largely subjective and may involve a large number of criteria.

In 2004, the Upper San Pedro Partnership (hereafter Partnership) in southeastern Arizona adopted Alley and others' (1999) definition as the foundation for a plan to achieve sustainable groundwater use for the Sierra Vista Subwatershed of the Upper San Pedro Basin (U.S. Department of Interior, 2005).

Beginning in the mid-1990s and continuing to today, the U.S. Geological Survey (USGS) has explored the hydrogeology and monitored and evaluated the groundwater and surface-water systems of the Sierra Vista Subwatershed. This included assuming the primary responsibility for producing an annual report to Congress (321 Report) evaluating Partnership efforts to achieve sustainable groundwater yield in the subwatershed (U.S. Department of the Interior 2005, 2006, 2007, 2008, 2011, 2012, 2013, 2014). Following the publication of the final 321 Report (U.S. Department of the Interior, 2014), the USGS along with a number of Partnership members (The Nature Conservancy, Bureau of Land Management (BLM), Cochise County, City of Sierra Vista, Fort Huachuca) requested that the USGS Arizona Water Science Center provide a comprehensive evaluation of the efforts to achieve sustainable groundwater use in the subwatershed. This report addresses that request.

Purpose and Scope

This report provides a comprehensive update on efforts to achieve sustainable groundwater use in the Sierra Vista Subwatershed. This is accomplished through the assessment of a suite of 14 indicators that are based on different parts of a broad, multiagency monitoring program. Sustainable use of groundwater in the subwatershed requires, at a minimum, a stable rate of groundwater discharge to and base flow in the San Pedro River. Most of the 14 indicators are therefore related to long-term or short-term effects on base flow and provide us with a means to evaluate where and how much rates of groundwater discharge to and base flow in the San Pedro River may be expected to change in the future.

The report evaluates general trends in groundwater, surface water, and water quality over the period of record, although the primary focus is on the ongoing period of intense groundwater and surface-water monitoring that began in the mid-1990s, and peaked in the 2000s. Some data were evaluated using statistical

methods, whereas other, more irregular datasets, were limited to visual inspection. With the exception of the water-quality data, increasing, stable, or decreasing indicator trends imply that increasing, stable, or decreasing discharge of groundwater, respectively, to the San Pedro River is occurring or is imminent.

The ongoing monitoring program in the subwatershed provides the underpinnings for the 14 indicators of sustainability evaluated in this report (table 1). Beginning in the early 2000s, a suite of partners (Agricultural Research Service (ARS), BLM, Cochise County, Fort Huachuca, Sierra Vista, The Nature Conservancy, and USGS) has provided financial and (or) in-kind support, which has allowed most of the monitoring to continue uninterrupted. The monitoring program components include USGS and ARS groundwater-level measurements at about 30 wells perforated in the regional aquifer and about 42 wells perforated in the San Pedro River alluvial aquifer, including seven deep/shallow pairs used to monitor vertical gradients. Streamflow is monitored at three USGS mainstem discharge gaging stations—San Pedro River at Palominas, AZ (09470500; Palominas), San Pedro River at Charleston, AZ (09471000; Charleston), and San Pedro River near Tombstone, AZ (09471550; Tombstone); one discharge gaging station on the Babocomari River—Babocomari River near Tombstone, AZ (09471400; lower Babocomari); and at one stage recorder station—San Pedro at Lewis Springs DCP (09470920; Lewis Springs). Streamflow permanence throughout the year is monitored by the mainstem gaging stations, the Lewis Springs stage recorder, and seven automatic digital cameras maintained and downloaded by ARS (formerly by BLM). Streamflow permanence along the entire riverine reach in the subwatershed is evaluated once per year, in June, during the wet-dry mapping campaign organized by The Nature Conservancy with assistance from BLM. Discharge is also measured by the USGS at three west side springs (Horsethief (313228110092701), Murray (313425110102301), and Moson (313624110101401)), one east side spring (Lewis (313456110081901), and one east side flowing well (McDowell-Craig (312502110060701).

As part of the USGS National Water-Quality Assessment Program (NAWQA), USGS has monitored a variety of San Pedro River water-quality constituents throughout the year at Charleston. Separately, the USGS Arizona Water Science Center analyzed the water quality of Murray Springs (2006, 2008, 2009, and 2010), and Lewis Springs and Horsethief Spring (2009), for emerging contaminants (pharmaceutical and wastewater constituents). However, this program has not continued. The USGS collects and analyzes stable-isotope samples from four gaging-station sites along the San Pedro River (Palominas, Lewis Springs, Charleston, and Tombstone), at the Lower Babocomari gaging station, and near the Hereford Bridge. In addition to four National Climate Data Center precipitation gages, ARS maintains a network of 21 additional precipitation gages across the subwatershed. Twenty-one soil moisture sensors are co-located with the precipitation gages. ARS also monitors evapotranspiration at the Charleston Mesquite monitoring site. Aquifer storage change is monitored in the subwatershed by the USGS using microgravity techniques (about 25 stations) and water budget accounting. There are also five mountain-based low-flow gaging stations, but they (and the ARS soil-moisture censors previously mentioned) were not used in the evaluation of sustainable groundwater use.

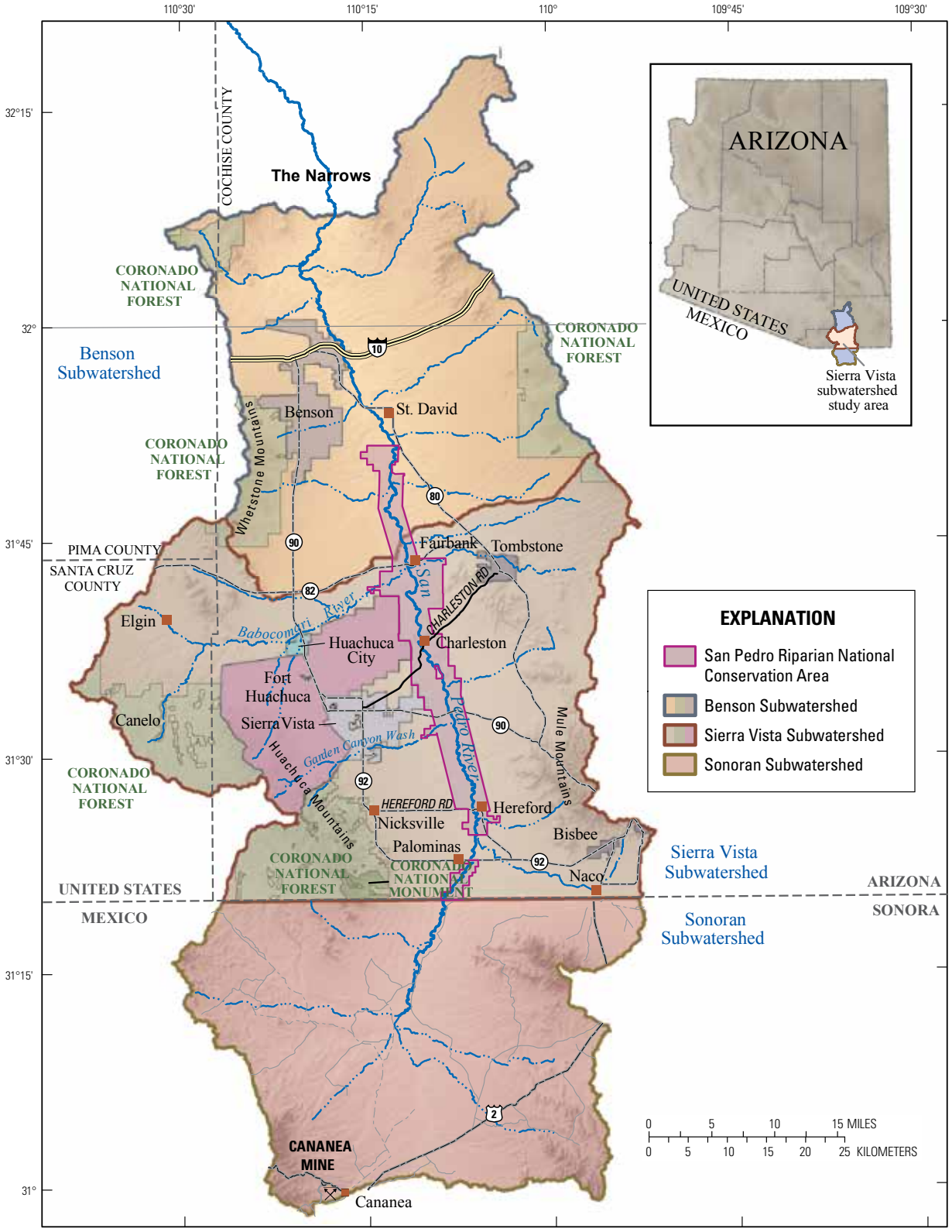
Description of the Upper San Pedro Basin and the Sierra Vista Subwatershed

The San Pedro River originates near Cananea in Sonora, Mexico, and empties into the Gila River near Winkleman, Arizona, 170 miles (mi) to the north. It is divided into an upper and lower basin separated by a bedrock constriction north of Benson, Arizona, commonly called “The Narrows” (Roeske and Werrell, 1973; S.G. Brown and B.N. Aldridge (unpub. data, 1973); Cordova and others, 2015). The Upper San Pedro Basin is subdivided into three subwatersheds. The Sonoran Subwatershed extends from the river’s source in Mexico to the international boundary; the Sierra Vista Subwatershed begins at the international boundary and terminates about 30 mi north at the Tombstone gaging station, 2 miles north of the ghost town of Fairbank and in the vicinity of the volcanic outcrops of the Tombstone Hills; the Benson Subwatershed extends from the Tombstone gaging station to The Narrows (Dickinson and others, 2010; Cordova and others, 2015).

The San Pedro River became entrenched along much of its length beginning in the late 1800s and was fully entrenched by about 1920 (Hereford, 1993; Huckleberry, 1996; Hereford and Betancourt, 2009). Entrenchment has led to the development of a narrow riparian forest—primarily cottonwood-willow (*Populus fremontii* and *Salix goodingii*)—in and adjacent to the entrenched channel throughout the length of the subwatershed and beyond. Adjacent to the riparian forest south of Charleston, retired agricultural lands dominate the pre-entrenchment alluvial surface; north of Charleston are dense mesquite forests. Above the pre-entrenchment flood plain is Chihuahuan desert scrub (Brown, 1982) and Chihuahuan desert grassland invaded by woody shrubs, principally mesquite. Mountain foothills include oak savannas, and at the higher elevations, conifers and aspens.

The 950 square miles (mi^2) Sierra Vista Subwatershed is part of a broad alluvial valley 4,000 to 5,000 feet (ft) above sea level. Located in the southern part of the Basin and Range physiographic province (Fenneman, 1928), it is bounded to the east and west by fault-block mountains. The Huachuca Mountains (~9,400 ft) and Mustang Mountains (~6,500 ft) are on the western perimeter of the subwatershed, whereas the Mule Mountains (~7,400 ft) and the southwestern edge of the Dragoon Mountains (~7,500 ft) are to the east. The Tombstone Hills (~5,300 ft) are within the subwatershed to the northeast. The San Pedro River itself is interrupted-perennial, with about 22 of the 35 river miles in the subwatershed perennial (Turner and Richter, 2011). The longest perennial reach is about 8 mi, beginning about a mile south of Arizona State Route 90 and ending about a mile north of Charleston.

The northern flank of the Sierra San Jose (~8,300 ft), just south of the international boundary, drains into Greenbush Draw, a major ephemeral tributary that reaches the San Pedro River north of Palominas (fig. 1). A second major ephemeral channel on the east side of the subwatershed is Walnut Gulch, which drains the



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

Coordinate System: NAD 1983 UTM Zone 12N; Projection: Transverse Mercator
Datum: North American 1983; False Easting: 500,000.0000
False Northing: 0.0000; Central Meridian: -111.0000
Scale Factor: 0.9996; Latitude Of Origin: 0.0000; Units: Meter

Figure 1. Map of the Upper San Pedro Basin, the Sierra Vista Subwatershed, and the San Pedro National Conservation Area, southeastern Arizona.

northeast part of the subwatershed in the vicinity of Tombstone, including the Tombstone Hills and a small part of the Dragoon Mountains. Nearly opposite where Walnut Gulch enters the San Pedro River is the interrupted-perennial Babocomari River, the largest tributary in the Upper San Pedro Basin. The Babocomari River drains the northwestern part of the subwatershed, including the northern end of the Huachuca Mountains, part of the Canelo Hills, and the Mustang Mountains. Walnut Gulch and the Babocomari River join the San Pedro River 2 to 3 mi upstream of the Tombstone gaging station. A large number of ephemeral tributaries empty into the San Pedro and Babocomari Rivers, with about twice as many on the west side of the subwatershed as on the east side (Coes and Pool, 2005).

Precipitation in southeastern Arizona is bimodal and falls preferentially on the higher elevations. About half of the annual subwatershed total comes during the summer rainy season (the North American Monsoon; Adams and Comrie, 1997), and about a third falls during the winter. Occasionally tropical systems can cause significant precipitation events in the autumn, and El Niño events often lead to wet winters and flooding, with February and March typically receiving the bulk of that precipitation (National Oceanic and Atmospheric Administration, 2016). In most years, there is a distinct fore-summer drought from late April through the end of June and a less distinct reduction in precipitation in middle to late autumn (Gungle, 2006).

The southwestern drought that began around 2000 and got underway in Arizona in 2002 (U.S. Department of Agriculture, 2015; National Drought Mitigation Center, 2015) may have had only a modest effect on average annual subwatershed precipitation (fig. 2). Four-station average precipitation showed no statistically significant trend from 1991 to 2012, although conditions appear to have been drier in the northern part of the subwatershed from 2002 to 2012 than in the southern part of the subwatershed (Hamerlynck and others, 2013). The 11-year subwatershed average precipitation for the years 1991 to 2001 was 15.36 inches (in.) and the 11-year average from 2002 to 2012 was 14.61 in. (National Oceanic and Atmospheric Administration, 2015; R.L. Scott, Agricultural Research Service, written commun., 2015; fig. 2). The seasonal timing of precipitation may have shifted, however, as evidenced by a decrease in years of total January to May discharge at the San Pedro River greater than 5,000 acre-feet (acre-ft) from the earlier 11-year period (4 years) to the later 11-year period (0 years) at Charleston, AZ, gaging station, and an increase in years of total June to August discharge greater than 10,000 acre-ft there over the same time periods (3 and 7 years, respectively; R.L. Scott, Agricultural Research Service, written commun., 2015). Cool season (December to April) precipitation at a site in Walnut Gulch, near Tombstone, shows a similar decrease from the 1990s to the 2000s (Hamerlynck and others, 2013).

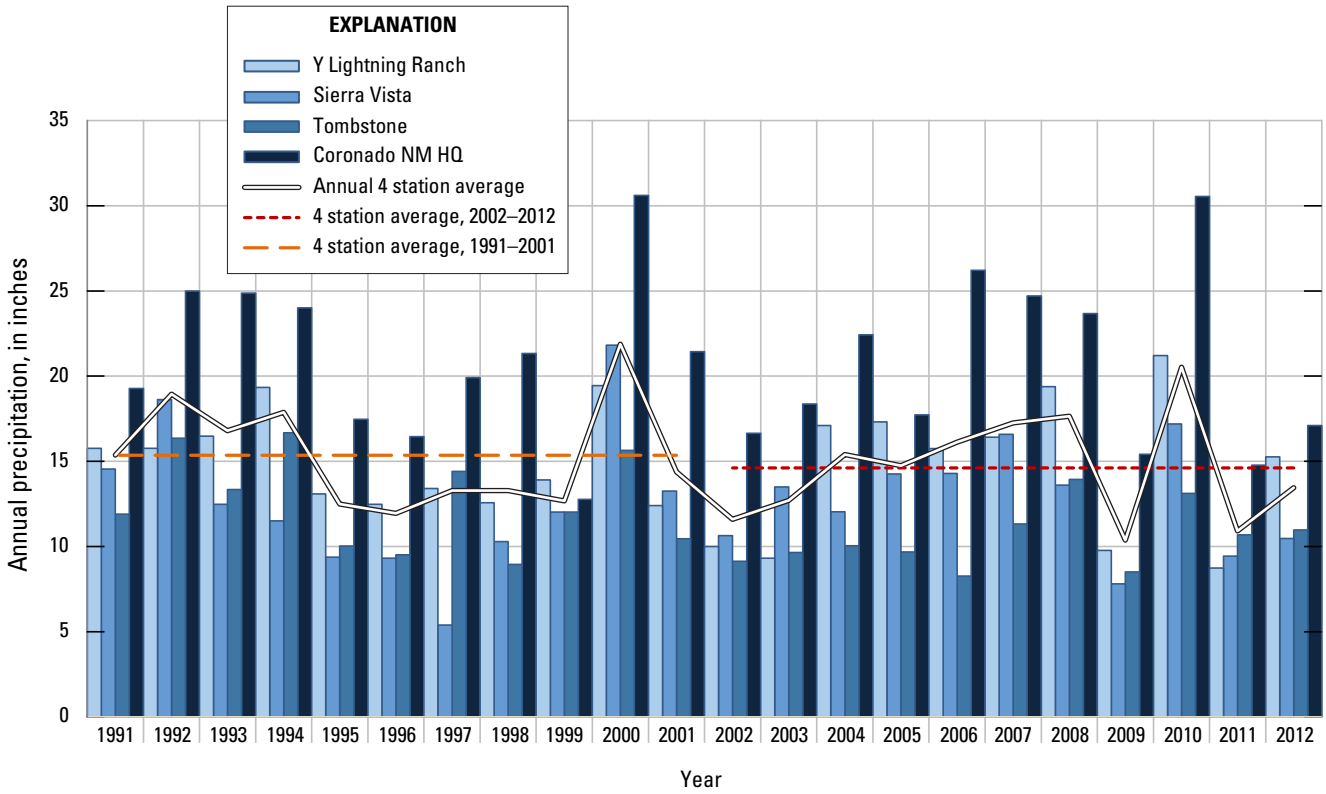


Figure 2. Graph showing annual precipitation record for four gaging stations in the Sierra Vista Subwatershed, southeastern Arizona, 1991 to 2012. Horizontal dashed lines are total average precipitation before and after the onset of the 2002–12 southwestern drought in Arizona.

Socioeconomic Setting

There were an estimated 80,800 people in the Sierra Vista Subwatershed in 2012 based on Arizona Department of Administration (2013) statistics. Sierra Vista is the largest city in the subwatershed, with an estimated 2012 population of about 45,800 people (an increase of 1,900 people from the 2010 census number; Arizona Department of Administration, 2013). The incorporated cities of Bisbee, Tombstone, and Huachuca City add another 8,600 people, and 26,500 people live in unincorporated areas of the subwatershed under the jurisdiction of mainly Cochise County (the westernmost part of the subwatershed includes small pieces of Santa Cruz and Pima Counties) (Arizona Department of Administration, 2013).

The region's largest employer is Fort Huachuca, which provided subwatershed residents with about 10,600 full time equivalent jobs in fiscal year (FY) 2011. The Fort's direct economic impact on the subwatershed is estimated to be about $908 million annually with a total economic output (direct, indirect, and induced) of around $1.7 billion (Maguire Company, 2008; Vernadero Group and Elliot D. Pollack and Company, 2009). The subwatershed's second largest employer is General Dynamics Info Technology with about 1,000 fulltime employees (FTE), followed by Sierra Vista Unified School District No. 68 with 770 FTE. Overall, government employees make up about 33 percent of the City of Sierra Vista workforce (Center for Economic Research, 2011).

In 2010, the cost of living in Sierra Vista and Cochise County was below the national average and the cost of living in the Sierra Vista-Douglas micro area was below that of all other major cities in Arizona except for Tucson. Like much of the Nation, Sierra Vista and Cochise County saw declines in many economic indicators beginning in 2008, including retail sales, new home permits, and median home price (Center for Economic Research, 2011).

The Upper San Pedro Partnership

The Upper San Pedro Partnership was created in 1998 through the Arizona Department of Water Resources' (ADWR) Rural Watershed Initiative, at least partially in an attempt to resolve widely varying opinions about the fate of the San Pedro River (Graham, 2007). Such a local advisory panel was also a key recommendation of the North American Free Trade Agreement's Commission on Environmental Cooperation (Davis, 2004). The Partnership replaced the earlier Cochise County Oversight Committee, as well as its Technical Review subcommittee. In addition, the Partnership provided a vehicle for Sierra Vista and Cochise County to work together alongside a range of Federal and State agencies, as well as with the other incorporated municipalities of the subwatershed and some nongovernmental organizations. According to its Web site, the Partnership's purpose is to meet the long-term water needs of both the San Pedro Riparian National Conservation Area (SPRNCA) and the area's residents (Upper San Pedro Partnership, 2013a). This is to be accomplished through the identification, prioritization, and implementation of policies and projects related to groundwater conservation and (or) enhancement. The Partnership has never reported to a particular jurisdiction, although Cochise County and Sierra Vista carry more weight in decision-making than other partners (Upper San Pedro Partnership, 2013b). In November 2003, the Partnership was recognized by the U.S. Congress through the Defense Authorization Act of 2004 (Public Law 108-136, Section 321) and charged with achieving sustainable yield of the subwatershed regional aquifer by September 30, 2011, as well as with publishing annual progress report updates.

Research and Assessment Into the Hydrologic Resources of the San Pedro River Valley and Sierra Vista Subwatershed

The earliest assessment of hydrologic resources in the San Pedro River valley was undertaken by Hill (1905) and followed later by Bryan and others (1934). Both focused on the Benson and St. David areas, although Hill (1905) also evaluated the suitability of the San Pedro River and adjacent lands near Charleston for a reservoir. Brown and others (1966) evaluated the water resources of Fort Huachuca. S.G. Brown and B.N. Aldridge (unpub. data, 1973) estimated San Pedro River surface discharge from the international boundary with Mexico to its confluence with the Gila River and inputs to the system from tributary inflow and mountain front recharge. Much of the assessment of hydrologic resources in the Upper San Pedro Basin that followed came as a byproduct of the development of groundwater models, including Freethey (1982), Vionnet and Maddock (1992), Corell and others (1996), Goode and Maddock (2000), and Pool and Dickinson (2007).

In November 1988, most of the San Pedro River within the subwatershed, including adjacent riparian areas, was protected by the U.S. Congress in Title I of Public Law 100-696, which created the San Pedro Riparian National Conservation Area (SPRNCA). It is managed by the Bureau of Land Management (BLM), which has been charged to do so ". . . in a manner that conserves, protects, and enhances the riparian area . . ." and the other resources found throughout the SPRNCA (Public Law 100-696). The Upper San Pedro Basin includes habitat for 389 avian species, 84 species of mammals, and 47 reptile and amphibian species (Steinitz, 2003). The SPRNCA includes about 40 mi of the San Pedro River riparian system, 30 mi of which is in the Sierra Vista Subwatershed (fig. 1).

Also in 1988, and then again in 2005, ADWR evaluated the groundwater resources of the Upper San Pedro Basin for Active Management Area (AMA) designation (Putman and others, 2008; Arizona Department of Water Resources, 2005a, 2005b; Arizona Revised Statutes 45-412). Jurisdictions within an AMA are required to actively manage groundwater through conservation and supply augmentation. Both the 1988 and 2005 evaluations by ADWR concluded that the

San Pedro Basin did not meet the statutory criteria for AMA designation (Arizona Department of Water Resources, 1988, 2005a, 2005b). Other State investigations included an ADWR (Arizona Department of Water Resources, 1990) inventory of San Pedro Basin surface water and groundwater subject to Federal surface-water rights as part of the Gila River adjudication. Around this same time Pool and Coes (1999) of the USGS described the state of the knowledge of the hydrogeology of the subwatershed and added to that an improved definition of the distribution of silt and clay layers in the region, an assessment of seasonal precipitation, runoff and base flow including the sources of base flow, and information on regional water-level changes.

The Defense Authorization Act of 2004 revised how Section 7 of the Endangered Species Act of 1973 (16 U.S.C. § 1531 et seq.) applies to the Fort Huachuca Military Reservation. Section 321 of the act excludes cumulative effects of water consumption, which are not related to Fort Huachuca, when determining whether the agency action is likely to jeopardize the continued existence of an endangered or threatened species or result in the destruction or adverse modification of designated critical habitat. As previously mentioned, Section 321 also recognized the Upper San Pedro Partnership, a consortium of 21 local jurisdictions, State and Federal agencies, and nongovernment organizations, and directed the Partnership to ". . . restore and maintain the sustainable yield of the regional aquifer [of the subwatershed] by and after September 30, 2011." Not stipulated in Section 321 is a definition of "sustainable yield of the regional aquifer," although the Partnership eventually adopted the definition of Alley and others (1999).

The push to achieve sustainable groundwater yield led to a number of Partnership-sponsored reports that either added to the body of hydrogeologic and biologic knowledge of the subwatershed or evaluated potential strategies and methods for achieving sustainable yield. Coes and Pool (2005) produced an assessment of ephemeral-stream channel and basin-floor infiltration and recharge, and Gungle (2006) analyzed the timing and duration of ephemeral streamflow in the subwatershed. Leenhouts and others' (2006) analyzed the hydrology, vegetation-hydrologic relations, and evapotranspiration (ET) requirements and plant-water sources in the SPRNCA. A statistical analysis of the trends in streamflow in the San Pedro River was also published by Thomas and Pool (2006), and the Cochise County Flood Control Urban Runoff Plan, which evaluated the size, placement, and efficacy of 30 proposed stormwater detention basins functioning as de facto recharge basins on the west side of the subwatershed was published by Stantec Consulting, Inc., and GeoSystems Analysis, Inc. (2006). Pool and Dickinson (2007) published a five-layer groundwater-flow model (two layers in the Mexican part) of the Upper San Pedro Basin, from the headwaters at Cananea in Sonora, Mexico, to just north of Fairbank, Arizona. This was followed by a capture and recharge analysis that used the new groundwater model to map the effects of pumping and recharge across the subwatershed on groundwater discharge to the riparian area (Leake and others, 2008). Kennedy and Gungle (2010) analyzed base-flow discharge from the subwatershed at the Tombstone gaging station. Most recently, Lacher (2011) updated the Pool and Dickinson (2007) groundwater-flow model and simulated the possible effects of population-growth-driven increases in pumping on groundwater levels through 2105 (Lacher and others, 2014). The possible effects of near-stream recharge on base flow was then simulated for three different sites on the San Pedro River through 2111 (Lacher 2012; Lacher and others, 2014).

Beginning in 2004, the Department of the Interior, represented by the USGS, in consultation with the Secretaries of the Departments of Agriculture and Defense and in cooperation with the Partnership published a series of annual reports entitled "Water Management of the Regional Aquifer in the Sierra Vista Subwatershed, Arizona—Report to Congress" (U.S. Department of Interior, 2005, 2006, 2007, 2008, 2011, 2012, 2013, 2014). This series of reports was mandated by Section 321 and required specific annual reporting related to the water budget balance.

In 2009, the Partnership chose to include in the annual "321" reports eight indicators of progress toward sustainable yield of groundwater in the subwatershed (U.S. Department of the Interior, 2011):

1. Regional-aquifer water levels
2. Near-stream alluvial-aquifer water levels
3. Near-stream vertical gradients
4. Base-flow on San Pedro and Babocomari Rivers (originally, days of no flow)
5. Streamflow permanence
6. Spring-flow discharge
7. Aquifer-storage change measured with microgravity
8. Water-budget balance

Additional discussion and evaluation of the available datasets by the Partnership's Technical Committee led to the inclusion of six additional indicators to assist in the evaluation of sustainable groundwater use:

1. Horizontal gradients (regional aquifer wells)
2. Annual fluctuation of near-stream alluvial-groundwater levels
3. June wet-dry status
4. San Pedro River water quality
5. San Pedro and Babocomari Rivers stable-isotope analysis
6. Springs water quality

Analysis, discussion, and evaluation of these 14 indicators of sustainable use of groundwater in the subwatershed forms the core of this report.

Sustainability and the Sustainable Yield of Groundwater

Sustainability has been conceptualized in two distinct ways (Farrell and Hart, 1998):

1. The critical limits view, which considers certain resources such as water as essential to both humans and ecological systems and that such resources constrain the Earth's population and its manner of living. Spoiled, destroyed, or otherwise wasted resources impact future populations and lifestyles;

2. The competing objectives view, which looks to balance or optimize resource use among different systems, typically environmental, economic, and social systems. This can be seen as attempting to meet a broad range of human needs and aspirations.

Although distinct, the two sustainability concepts are compatible—the resource requirements of the environment and economic and social systems today are balanced not just against each other but also against the same needs in the years to come.

History of Sustainable Yield

To place the goal of sustainable groundwater yield in context, it is useful to consider the history of sustainable yield, which derives from the earlier concept of "safe yield." Alley and Leake (2004), Kalf and Woolley (2005), and Zhou (2009) all provide reviews of the development of the sustainable yield concept from its origins as safe yield.

Defined by Lee (1915), safe yield is "the limit to the quantity of water which can be withdrawn regularly and permanently without dangerous depletion of the storage reserve." In particular, Lee's observations pertained to closed basins of the arid Southwest. For large aquifers, Lee allowed for periodic withdrawal from storage in times of drought, but observed that, in general, the safe supply available for pumping is equal to the amount recharged minus unpreventable residual losses (subflow out of the basin, low-elevation springs, artesian waters, and rejected recharge). Lee (1915) also observed that "a basin has not been fully developed as long as the evaporating area [surface-water body] persists in times of drought," and that most losses from artesian waters can be "dried up" by heavy pumping.

Lee's (1915) observations are important for a number of reasons. First, they established that the rates of natural inflow (recharge) and outflow (discharge) from a groundwater basin are in equilibrium. Second, they recognized that groundwater discharge can be captured by pumping. Third, they recognized that the safe yield of a basin will generally be less than the amount of natural recharge due to unpreventable residual losses that cannot be captured by pumping. Fourth, they underscore that the goal of safe yield in 1915 was to ensure that the groundwater resource provided the maximum supply possible to support commercial and residential development and then to maintain that indefinitely. From a critical limits standpoint, the focus was strictly on socioeconomic sustainability; any surface water left in the basin was considered an unpreventable, wasted, residual loss. From a competing assets standpoint, acceptable environmental consequences in 1915 included the loss of all surface water and nonagriculture vegetation from the basin.

Meinzer (1923) defined safe yield as a rate of groundwater withdrawal—specifically for human use—that will not deplete the supply to such an extent that withdrawal at the given rate becomes economically infeasible. Meinzer (1932) clarifies this definition to be an equilibrium condition in the aquifer, ". . .where there is no further net withdrawal from storage and no further permanent lowering of the water table." He notes that "salvage"—what we would call "capture" today—of natural discharge can continue to increase until all pumping comes entirely from salvage/capture, so long as safe yield/equilibrium is not exceeded. Meinzer (1932) clearly recognized that some natural discharge would continue to occur even in the face of heavy pumping.

The goals for water use remained largely unchanged into the 1940s, as Meinzer (1932) shared the general perspective of Lee (1915) that natural discharge cannot be wholly prevented while implying it is wasted water. Like Meinzer (1932), Theis (1940) recognized the practical difficulty of capturing all the natural discharge and rejected recharge in a basin and so recommended that pumps be placed as close as possible to areas where "non-productive vegetation" or groundwater-fed surface water was being lost. He noted that if rejected recharge is being discharged by transpiration from "non-beneficial vegetation, no economic loss is suffered" by capturing this discharge. Societal values regarding native vegetation and related wildlife habitat have clearly changed in the intervening years and with it the constraints on groundwater development. This is reflected in the shift from a focus on safe yield, which has come to be seen as overlooking or ignoring the value of natural systems, to one of sustainable yield, which does not overlook the value of natural systems.

Theis (1940) also made a number of important observations. First is the fundamental observation that water discharged by wells must be balanced by a loss somewhere, and some of this loss will always include groundwater storage—some groundwater will always be mined. Second, a reduction in natural discharge or increase in recharge (induced from a surface-water source) results from increasing the pressure gradient between a well and the area of natural discharge or potential recharge, that is, by expanding the cone of depression to where it reaches these areas. Third, once pumping commences, a new equilibrium cannot be established until sufficient time has elapsed for the cone of depression to reach the areas of natural discharge and potential recharge, and equilibrium will only then be reached if the total amount pumped does not exceed the total amount of natural discharge and induced recharge available for capture. In other words, capture of natural discharge (and rejected recharge if any) must offset pumping to achieve equilibrium.

Beginning in 1952, Lohman started defining safe yield as "the amount of ground water one can withdraw without getting into trouble," where "trouble" is defined as "almost anything under the sun" (Lohman, 1972). Similarly, Todd (1959) defined safe yield as, "the amount of water that can be withdrawn [from

an aquifer] annually without producing an undesirable result." The effect of these more qualitative definitions is to both remove the quantitative restraints implicit in Theis' (1940), Meinzer's (1932), and even Lee's (1915) groundwater development goals—to not pump past equilibrium conditions—and at the same time to allow for the possibility that the effects of pumping may cause problems unrelated to the economics of groundwater development even before equilibrium conditions are achieved. In other words, in the first case Lohman's and Todd's definition allows for safe yield to include ongoing aquifer storage depletion if declining water levels are not an undesirable result for those affected. In the second case—and on the other hand—their "safe yield" can limit pumping to amounts well above equilibrium conditions if the effects of pumping are causing undesirable results unrelated to groundwater levels in wells, such as impacts to surface waters or riparian systems.

In the second edition of his text, Todd (1980) revised his terminology from "safe yield" to "perennial yield" (the definition itself is left largely unchanged) because:

> In the past the term "safe yield," implying a fixed quantity of extractable water basically limited to the average annual basin recharge, has been widely used. The term has now fallen into disfavor because a never-changing quantity of available water depending solely on natural water sources and a specified configuration of wells is essentially meaningless from a hydrologic standpoint.

Although Lee (1915), Meinzer (1932), and Theis (1940) recognized that safe yield was dependent on the amount of capture, and Theis (1940) and Meinzer (1932) clearly recognized that it was impractical to expect to capture all discharge and potential recharge in a basin, others in the interim had incorrectly simplified the safe yield concept to mean the amount of pumping in a groundwater basin that is *equal* to the amount of natural recharge. In nearly every case, pumping equal to recharge will result in a perennial overdraft situation on the basin scale equal to the residual losses such as underflow out of the basin, ET, and base flow—in other words, any uncaptured discharge from the basin.

Bredehoeft and others' (1982) paper, "Groundwater: The Water Budget Myth," emphasized the point that pumping equal to recharge within a groundwater basin will result in groundwater mining; sustained (continued) groundwater yield is limited by what discharge and induced recharge can be captured, not by the amount of precipitation that recharges the aquifer. Additionally, they observed that not only does the response of the groundwater system depend on the aquifer parameters and the boundary conditions, but also the positioning of the groundwater development within the system, an observation also made by Theis (1940). The rate at which natural discharge can be captured is dependent on well placement—the water levels in one part of the basin may decline drastically and wells can go dry long before enough capture has occurred elsewhere to establish a basinwide equilibrium state. Bredehoeft and others (1982) conclude that the "[m]agnitude of (groundwater) development depends on hydrologic effects that you want to tolerate," and that a groundwater budget does not help to determine this; it is dependent entirely on the amount of discharge the development can capture, and thus whether equilibrium can be established in a timely manner.

By the late 1980s, the term "sustainability" had entered the mainstream. In the field of groundwater, Bredehoeft and others (1982) had laid the groundwork and provided the vocabulary for subsequent discussions of sustainable groundwater yield. In a 1997 editorial in the journal Groundwater, Marios Sophocleous brought attention to the fact that safe yield does not lead to sustainable yield. Even when properly applied, safe yield allows for pumping equal to the capture of all groundwater discharge possible from a groundwater basin (Sophocleous, 1997). However, not all appropriable supply within a basin can necessarily be captured, which leads to continued water-level declines and continued storage depletion (Theis, 1940; Konikow and Leake, 2014). Additionally, capture of natural discharge includes springs, stream base flow, and ET. However, streams and springs are often depleted and vegetation has died off long before pumping reaches safe yield, regardless of definition (Bredehoeft, 1997). Sophocleous (1997) realized that "(w)e can maximize our (safe yield) of water by drying up our streams, but when we do, we learn that the streams were more than just containers of usable water"; they have value to society beyond that of a marketable commodity alone. The definition of "groundwater sustainability," therefore, reaches beyond just hydrologic systems and includes riparian and aquatic ecosystems. Still, a quantitative methodology for estimating a sustainable yield of groundwater has yet to be perfected (Sophocleous, 1997).

Alley and others (1999) produced USGS Circular 1186 on groundwater sustainability, including the definition adopted by the Partnership (". . .use of ground water in a manner that can be maintained for an indefinite time without causing unacceptable environmental, economic, or social consequences"). Alley and others (1999) also acknowledged that what may be an acceptable amount of groundwater withdrawal in terms of a groundwater system may be unacceptable with regard to a surface-water resource it affects (captures). The definition of sustainable yield continues to trend in this direction today—a groundwater withdrawal is sustainable if the effects of that withdrawal are acceptable to all who are or who will somehow and in any way be affected by it. For example, the California Water Foundation (2014) recently defined sustainable groundwater management as,

> . . . the management of a groundwater subbasin to provide for multiple long-term benefits without resulting in or aggravating conditions that cause significant economic, social, or environmental impacts such as long-term overdraft, land subsidence, ecosystem degradation, depletions from surface-water bodies, and water quality degradation, in order to protect the resource for present and future generations.

Thus, today's definition of groundwater sustainability has its roots in Lohman's (1972) and Todd's (1959) definitions of safe yield (the amount of water that can be pumped without

"causing trouble" or "producing an undesirable result"). There is no single, correct formula to derive some accurate sustainable yield value (Maimone, 2004); rather, sustainability is a "value-laden concept and one that in many respects is in the eye of the beholder."

Current Definition of Sustainable Groundwater Use

Many consider a qualitative definition of sustainable groundwater use to be unworkable for the purpose of evaluating the impact of groundwater withdrawals. To this end, there have been some recent attempts to quantify sustainable groundwater withdrawals (for example, Kalf and Woolley, 2005) and to assess sustainability within a probabilistic framework ("a sustainable system (is) one that maintains acceptable risks over an indefinite time horizon"; Howard, 2002). As a matter of practical application, sustainable yield is sometimes quantified as a percentage of natural and artificial recharge. Ponce and Bakoobehi (2010), for example, place sustainable use at somewhere between 10 and 70 percent of recharge, with average values at about 40 percent. Of course, this is in contradiction to Bredehoeft and others' (1982) observation that it is available capture, not recharge, on which sustainable pumping rates should be based; available sources of capture in a basin may not be adequate to offset pumping rates (Bredehoeft and others, 1982; Konikow and Leake, 2014). However, Kalf and Woolley (2005) and Zhou (2009) argue that Bredehoeft and others (1982), Bredehoeft (1997), and Bredehoeft (2002) overstate the need to base sustainable withdrawal on capture alone. In their opinion, natural recharge is still important when scaling groundwater development.

The difference in opinion may lie in the size of the development and distribution of the wells—Kalf and Woolley (2005) and Zhou (2009) may be looking at pumping evenly distributed across a theoretical basin, whereas Bredehoeft (1997) is considering the effects of pumping more locally, where a well field can drive water levels below adjacent well screens and (or) dry up a proximate stream or pond well in advance of reaching basinwide equilibrium. However, as Bredehoeft and others (1982) and Konikow and Leake (2014) show, pumping can only reach basinwide equilibrium—the baseline requirement for sustainability—if and when it is offset by sources of capture. As a result, the amount of pumping an aquifer can tolerate will not be known until the available sources of capture are known and quantified, and it may not be possible to determine this in advance. At local scales, of course, the importance of basin-scale natural recharge and of basinwide sources of capture on the sustainability of a given surface-water source is small in comparison to the importance of pumping location. From this perspective, sustainability of groundwater use is measured by the effect pumping has on specific surface-water and riparian features and by society's willingness to accept those effects, as discussed above.

There is general consensus that sustainable yield of groundwater must be defined from the perspective of constraints. For example, Evans and Collins (2010) define sustainable groundwater pumping as, "the volume left over after non-extractive groundwater needs have been satisfied," which implies that nonextractive (mostly environmental) needs constrain the availability of groundwater for other uses (mostly societal and economic). More typically, the three competing needs—environmental, societal, and economic—are components of a dynamic system, set in an equal opposition, linked to and constrained by the others. Of course, uses can overlap, and thus each is not wholly opposed to the other two. Economically driven groundwater use can also fulfill societal needs (for example, water companies). Water used for environmental needs (for example, maintain a riparian area, aquatic habitat) can also support economic (for example, tourism) and societal needs (for example, recreation, downstream water rights). Also, although each of the three needs requires groundwater, environmental needs typically require that the water be left in place, whereas economic and societal needs more often require it be extracted. Thus, there are often direct costs associated with the economic and societal uses that may not exist for environmental use.

Nonetheless, to achieve or maintain sustainable use, each of the three competing assets—the environment (water needed by ecosystems, including the plants and animals found there), the economy (water needed by industry, such as for mining, manufacturing, irrigation), and society (water needed by people to live, such as to drink, cook, and bathe)—must be kept in balance. In other words, they can use groundwater up to a point just short of causing unacceptable adverse impacts to the other two assets and while ensuring that groundwater for all three remains plentiful into the future.

Currently in the Sierra Vista Subwatershed there are no serious constraints on water availability for economic and societal needs, although this could change in the coming decades. Groundwater available for the environmental needs of the subwatershed, especially for discharge to the San Pedro River (base flow), has been declining since at least the 1930s (Pool and Coes, 1999). Sustainable use of groundwater is thus dependent, at a minimum, on stabilizing groundwater discharge into the San Pedro River. As a means to assess to what degree this may be beginning to happen, it is necessary to evaluate a suite of indicators of sustainability—many of which are related to short-term or long-term effects on base flow.

Indicators of Sustainable Groundwater Use in the Sierra Vista Subwatershed

Indicators provide evidence about the condition of a system and (or) of how it has changed (Bertram and Stadler-Salt, 2000; Vrba and Lipponen, 2007; Strange and Bayley, 2008). However, evaluation of indicators can be challenging and generally requires stakeholder discussion and decision. Stakeholders must agree on the tipping point for each indicator. A tipping point is typically an indicator value that is understood to show when a situation or set of conditions have changed from being sustainable to being unsustainable, or vice versa. Reaching such stakeholder agreement on indicator tipping points can be understandably difficult.

Sets of indicators developed to evaluate sustainable use of groundwater are not common. Much work with environmental

indicators has been done in the Great Lakes region (Lake Superior Binational Program, 1998; Bradof and others, 2000; Bertram and Stadler-Salt, 2000), and groundwater indicators have been proposed and developed for Michigan (Watkins and others, 2004; Steinman, 2007). Groundwater indicators have also been developed for the international community (Vrba and others, 2005; Vrba and Lipponen, 2007; Lavapuro and others, 2008), where to date they have been applied in case studies to Seville, Spain; Finland; Sao Paulo, Brazil; and South Africa (Vrba and Lipponen, 2007; Lavapuro and others, 2008). To our knowledge, the Partnership appears to be one of the first organizations to work toward sustainable groundwater yield at a subwatershed level.

In 2009, the Partnership Technical Committee developed a set of eight indicators based mostly on data available from an ongoing subwatershed monitoring program (U.S. Department of the Interior, 2011). Indicators for a limited region such as the subwatershed can have greater specificity than those of, for example, Vrba and Lipponen (2007), because a very specific environmental concern—the health of a specific riparian and aquatic system—is being addressed. Because Vrba and Lipponen's (2007) indicators are intended for regional and national assessment, they tend to be broad in their application and focused on groundwater supplies generally. The more specific, localized indicators used to assess sustainable groundwater yield in the subwatershed are keyed to specific groundwater and surface-water concerns found there. In the subwatershed, decreases in groundwater levels of just a few feet could be the difference between health and degradation of the San Pedro River aquatic system, near-stream herbaceous plant groups, and the cottonwood-willow riparian forest (Stromberg and others, 2006; Horton and others, 2001; Scott and others, 1999, 2000; Shafroth and others, 2000).

Hydrological Monitoring and Environmental Indicators in the Subwatershed

Comprehensive monitoring of groundwater and surface-water resources in the subwatershed has been taking place since the mid-to-late 1990s, and provides the basis for the hydrological indicators. Stromberg and others (2006) divided the SPRNCA into 14 reaches from south to north and then conducted a biohydrological analysis and a functioning-condition assessment of each reach. On the basis of that analysis and assessment, they assigned each reach a condition class (dry, intermediate, wet). Four of the Partnership's eight indicators follow Stromberg and others' (2006) work, providing updated information on the functioning condition of most of the reaches—near-stream alluvial aquifer water levels, near-stream vertical gradients, base-flow discharge, and streamflow permanence. Three of the remaining Partnership indicators relate to changes in regional aquifer storage—regional-aquifer water levels, aquifer storage change from microgravity measurements, and the annual groundwater budget balance. The remaining Partnership indicator, springs discharge, provides point-source data related to aquifer groundwater discharge.

Table 1 includes six additional indicators beyond the eight used by the Partnership in the 321 report. The new indicators make use of existing data that have not previously been incorporated into the Partnership's annual reporting and include horizontal groundwater gradients, surface-water isotope analysis (provides information on any changes to source of water in the stream), mainstem dry-season wet-dry status, and water quality. Combined, there are a total of 14 indicators of sustainable groundwater yield for the subwatershed. These have been grouped for discussion and evaluation into four groups based on their physical relation to the San Pedro riparian system (table 1). Group 1 includes indicators that evaluate sustainable yield from a subwatershedwide perspective (4 indicators). Group 2 indicators are focused on the alluvial aquifer adjacent to the San Pedro River (3 indicators). Group 3 indicators are related to the river itself (5 indicators). Group 4 indicators are based on characteristics of springs in the vicinity of the riparian area (2 indicators).

In the remainder of the report, each of the 14 indicators is discussed first in a methods section and then in a results section. The methods section defines indicators, describes their characteristics, and then details how monitoring records for the indicator were evaluated. The results section provides the monitoring data for the period of record, evaluates the trends in that record, if any, and discusses possible causes driving the trends. This is followed by a general discussion of overall indicator trends and implications for sustainable groundwater yield in the subwatershed.

Table 1. Indicators of sustainable groundwater use for the Sierra Vista Subwatershed, southeastern Arizona.

No.	Indicator group
	Group 1, Subwatershedwide indicators
1	Regional-aquifer water levels
2	Horizontal gradients (regional-aquifer wells)
3	Aquifer-storage change measured with microgravity
4	Annual groundwater-budget balance
	Group 2, Riparian system indicators
5	Near-stream alluvial-aquifer water levels
6	Near-stream vertical gradients
7	Annual fluctuation of near-stream alluvial-aquifer water levels
	Group 3, San Pedro River indicators
8	Streamflow permanence
9	Base-flow on San Pedro and Babocomari Rivers
10	June wet-dry status
11	San Pedro River water quality
12	San Pedro and Babocomari Rivers isotope analysis
	Group 4, Springs indicators
13	Springs discharge
14	Springs water quality

Group 1. Subwatershedwide Indicators

Indicator 1. Regional-Aquifer Water Levels—Methods

Regional aquifer water levels were measured in 30 nonpumping wells across the subwatershed that are screened in the basin fill aquifer (fig. 2; table A5). Fort Huachuca, USGS, and ARS have monitored these wells since the late 1990s. All but one of the regional aquifer wells (Palominas Deep) are located outside of the riparian area, and about half of these wells are located on Fort Huachuca (TW and MW wells). Beginning in the mid-1990s, most wells have been measured at least annually and in many years quarterly. Water levels are measured manually to one hundredth of a foot, although measuring tape uncertainty and other factors reduce the accuracy somewhat (Tom Porter, oral communs., 2014, 2015). A subset of the wells is instrumented with automatically recording pressure transducers, which are calibrated and adjusted based on manual tape measurements. USGS performs quality assurance, including data review for errors and inconsistencies, on all data before entry into the USGS National Water Information System (NWIS) database, regardless of agency making the measurement. Over the period of record, water levels across the subwatershed changed tenths of a foot to tens of feet.

Water levels in 28 of the regional aquifer wells were investigated for trends. Declining regional-aquifer water-level trends imply a deepening cone of depression and declining gradients which, if the trend were to continue, would lead to streamflow capture and reductions in groundwater discharge to the San Pedro River (Barlow and Leake, 2012). Increasing regional-aquifer water-level trends would imply increasing gradients and an eventual increase in groundwater discharge to the San Pedro River.

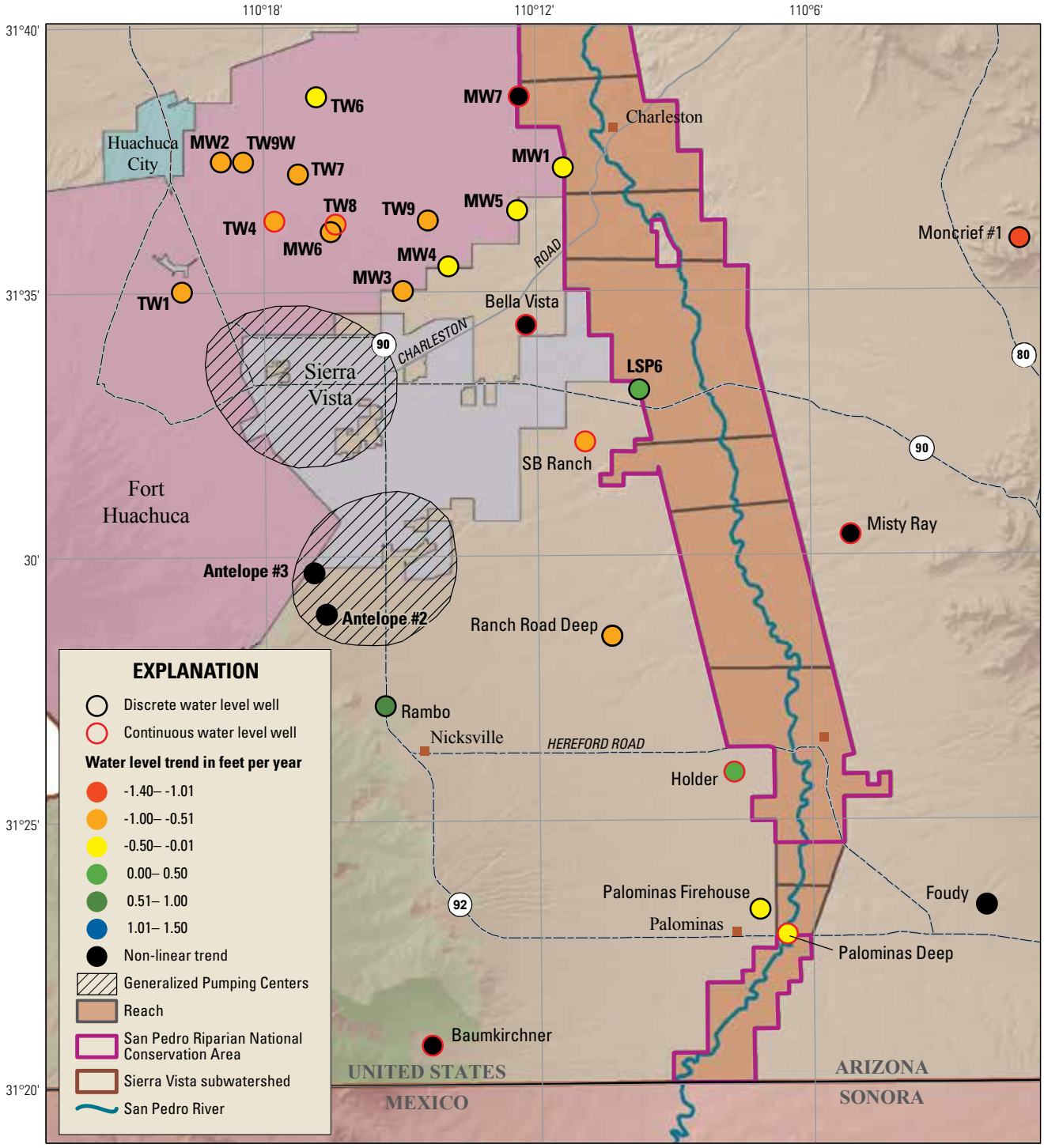
Additional groundwater data that include measurements from hundreds of pumping wells (not pumping at the time of measurement) are available from ADWR (Schmerge and others, 2009; Barnes and Putman, 2003; Barnes, 1997) and USGS (Konieczki, 1980). Water levels were assessed for linear trends using the methods of Tillman and others (2008) and a coefficient of determination (R^2) of 0.8 or greater. Hydrographs that do not satisfy this criterion most likely still contain hydrologic information such as seasonal changes (monsoon recharge) or depletion and recovery (in response to retirement of agricultural pumping) that, although useful, may be more difficult to interpret unambiguously than a simple linear trend.

Indicator 1. Regional-Aquifer Water Levels—Results

Eighteen wells had downward trending water levels (fig. 3), and the majority of these were on Fort Huachuca. Except along the eastern perimeter, declines ranged from 0.4 and 0.7 foot per year (ft/yr) going back as far as the early 1970s (for example, MW2, MW3, TW7, and TW8 in fig. 4). Water levels in the easternmost wells most distant from the pumping centers and closest to the SPRNCA are declining at rates of 0.10 to 0.15 ft/yr (for example, MW1 and MW7 in fig. 5).

The regional wells located off of Fort Huachuca showed a variety of trends, including short-term fluctuations superimposed on longer term increases (for example, Holder, LSP6, fig. 6), decreases over relatively short periods of record (for example, Palominas Firehouse, Ranch Road Deep, SB Ranch, Moncrief #1, fig. 7) and longer-term decreases (Bella Vista, Palominas Deep, fig. 8). Water levels in the Holder well, located west of San Pedro River reach 2 (fig. 7), rose several feet and lost much of their previous seasonal fluctuation after nearby agricultural pumping was discontinued in the mid-2000s.

Changes in other wells correspond to multiyear climatic cycles of recharge and drying, such as in the Antelope Run #3 well in Garden Canyon Wash near the Huachuca Mountains and the Foudy well, north of Highway 92, near the Mule Mountains and adjacent to a wash (fig. 9). These regional-aquifer wells have similar hydrographs, although the variation in water levels in Antelope Run #3 is more than an order of magnitude greater than in Foudy. The Ranch Road Deep and SB Ranch well are in the area of large-lot development and exempt residential well pumping southeast of Sierra Vista, and this may explain the steady decline in water levels in that area (fig. 7).



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: -111.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Figure 3. Map showing locations of regional-aquifer wells and respective groundwater-level trends in the Sierra Vista Subwatershed, southeastern Arizona. Negative values indicate declining water-level trends. (Trends were not evaluated for statistical significance; "non-linear trend" describes variable water-level patterns that cannot be readily described by a straight line.)

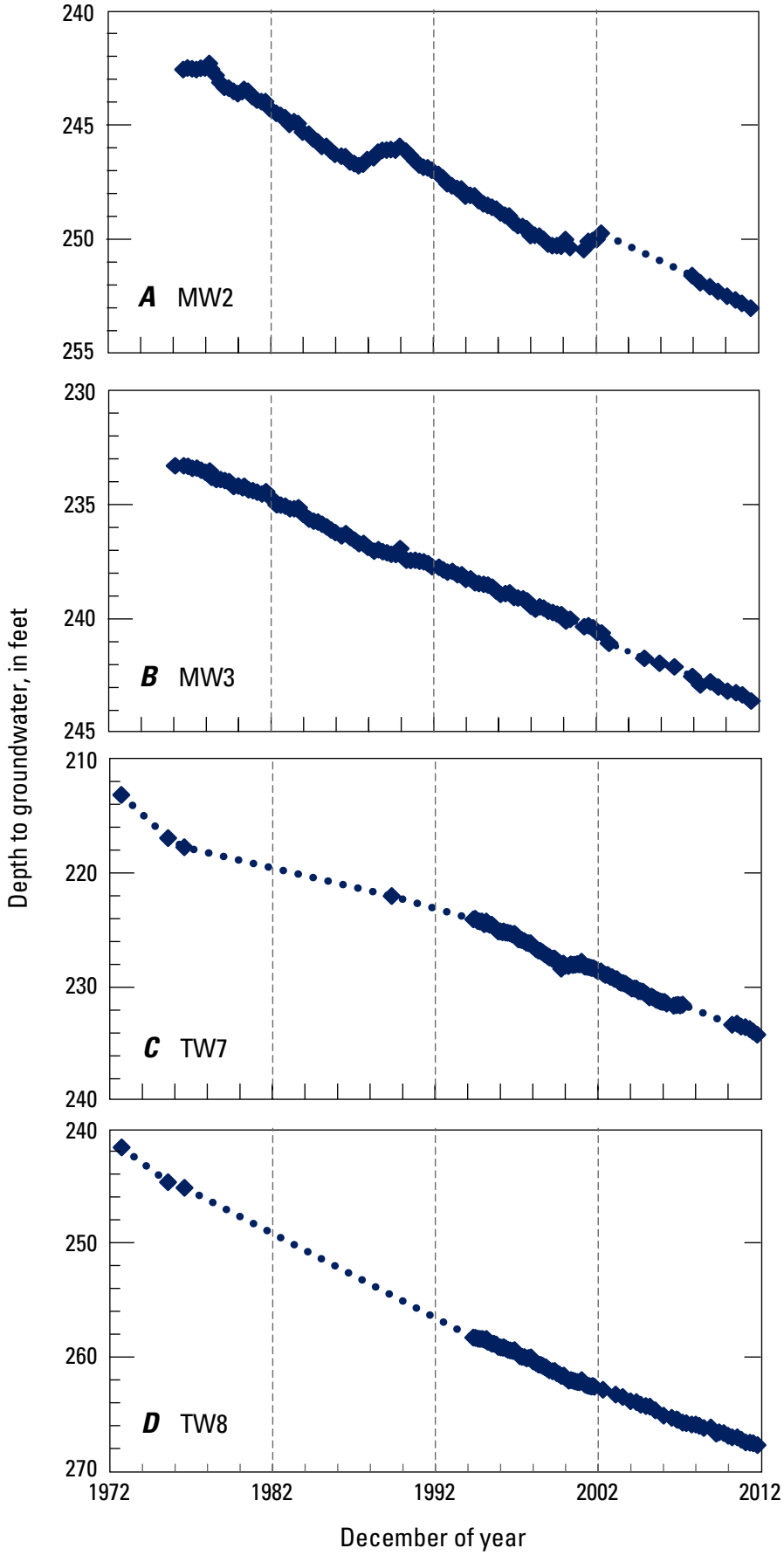


Figure 4. Graphs showing regional-aquifer water levels measured at four of the westernmost monitoring wells (MW2, MW3, TW7, and TW8) on Fort Huachuca, Arizona. All water levels measured in this area have fairly consistent declines. See figure 3 for well locations.

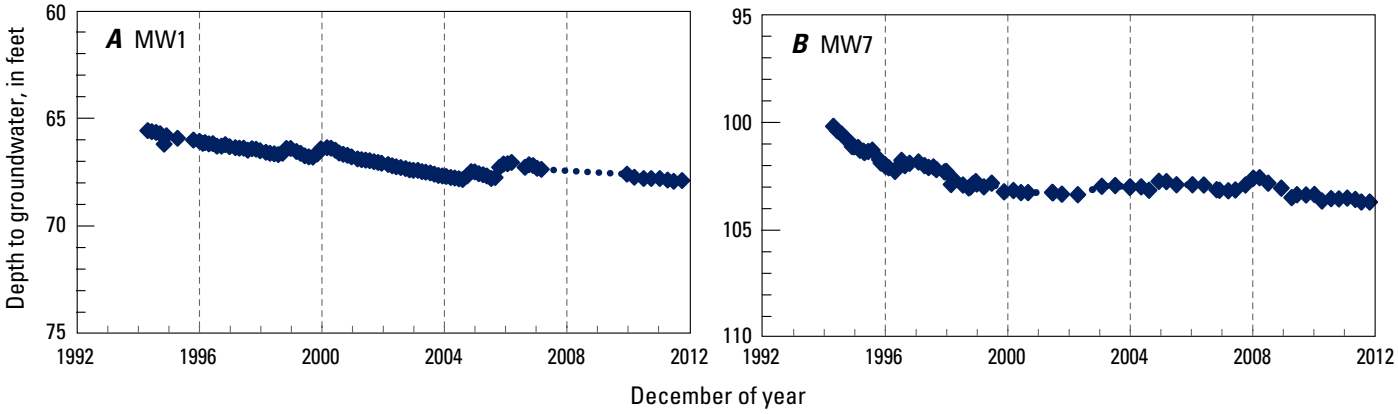


Figure 5. Graphs showing regional-aquifer water levels measured at two of the easternmost monitoring wells (MW1 and MW7) on Fort Huachuca, in the Sierra Vista Subwatershed, southeastern Arizona, beginning in the early 1990s. Water-level declines in these wells are smaller and less consistent than water-level declines in wells further west. See figure 3 for well locations.

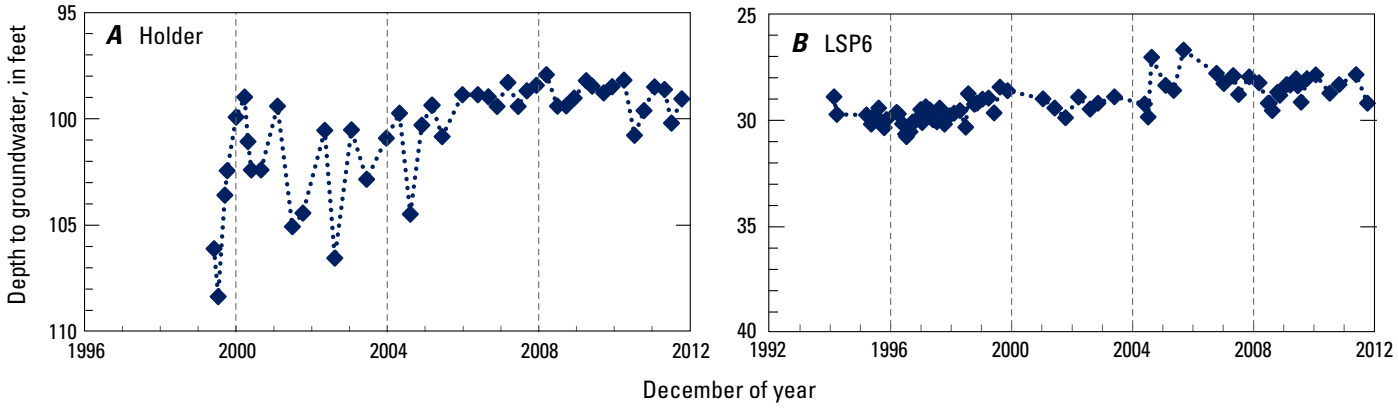


Figure 6. Graphs showing two regional alluvial-aquifer water levels (wells Holder and LSP6) that show an increasing trend in the Sierra Vista Subwatershed, southeastern Arizona. Agriculture pumping nearby to Holder ceased in the mid-2000s. See figure 3 for well locations.

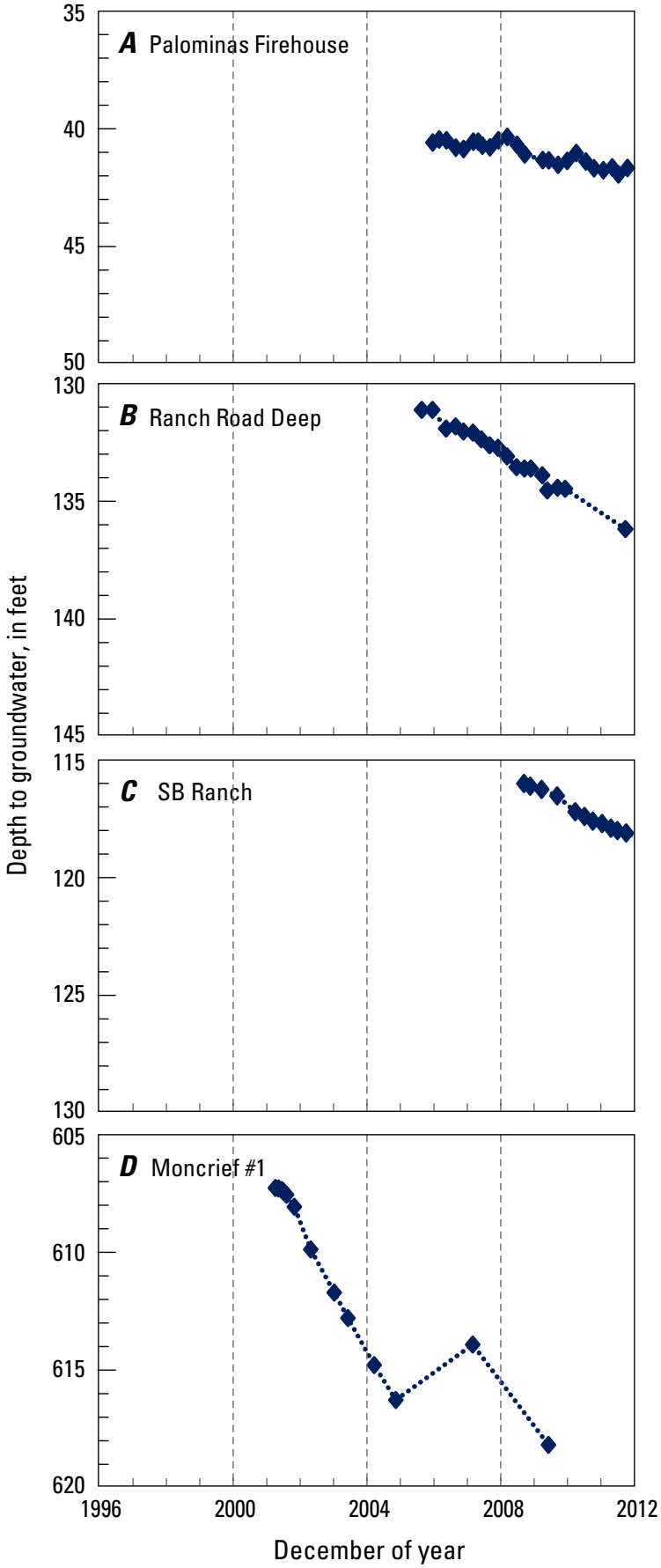


Figure 7. Graphs showing four regional aquifer water levels (wells Palominas Firehouse, Ranch Road Deep, SB Ranch, and Moncrief #1) that show relatively recent decreasing trends in the Sierra Vista Subwatershed, southeastern Arizona. Palominas Firehouse, Ranch Road Deep, and SB Ranch wells are south of Sierra Vista, west of the San Pedro River; Moncrief #1 is in the Tombstone Hills east of the San Pedro River. See figure 3 for well locations.

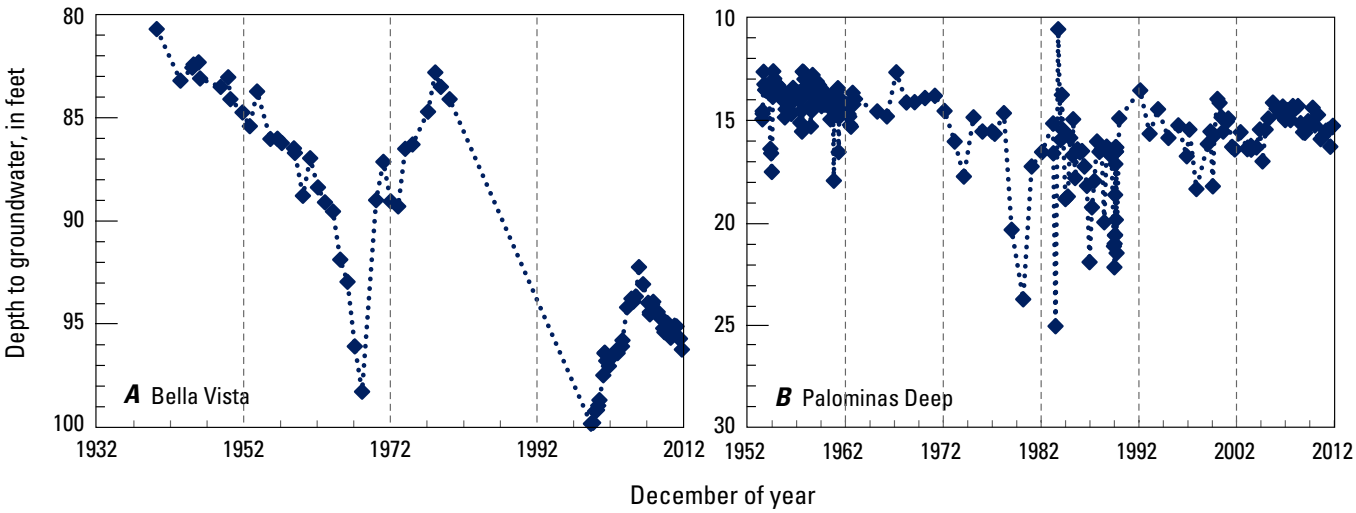


Figure 8. Graphs showing two regional-aquifer water levels (wells Bella Vista and Palominas Deep) that show a variable but overall long-term decline in the Sierra Vista Subwatershed, southeastern Arizona. See figure 3 for well locations.

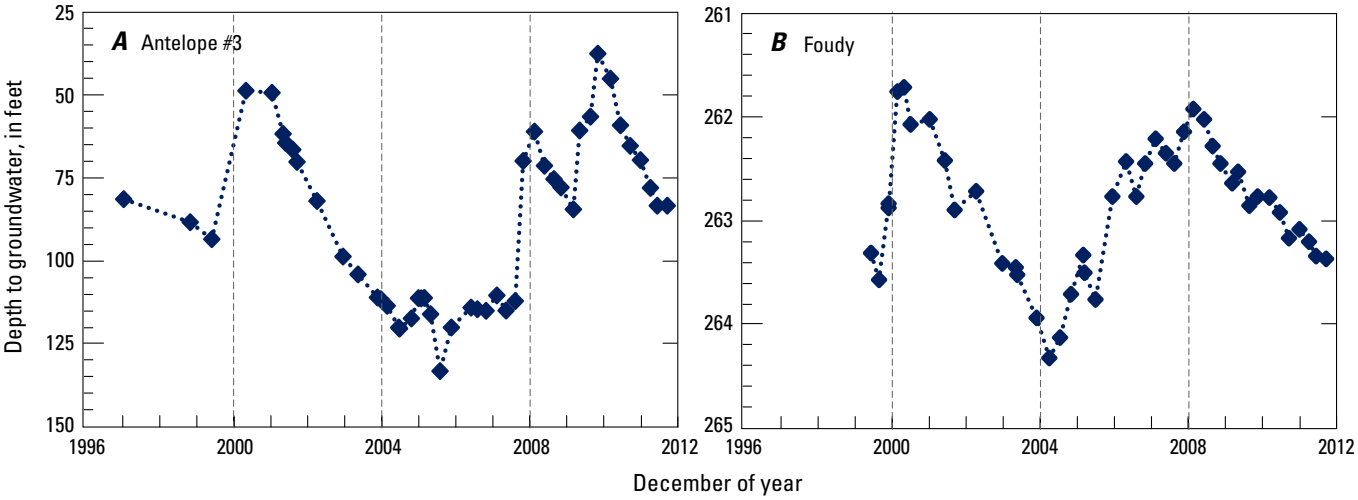
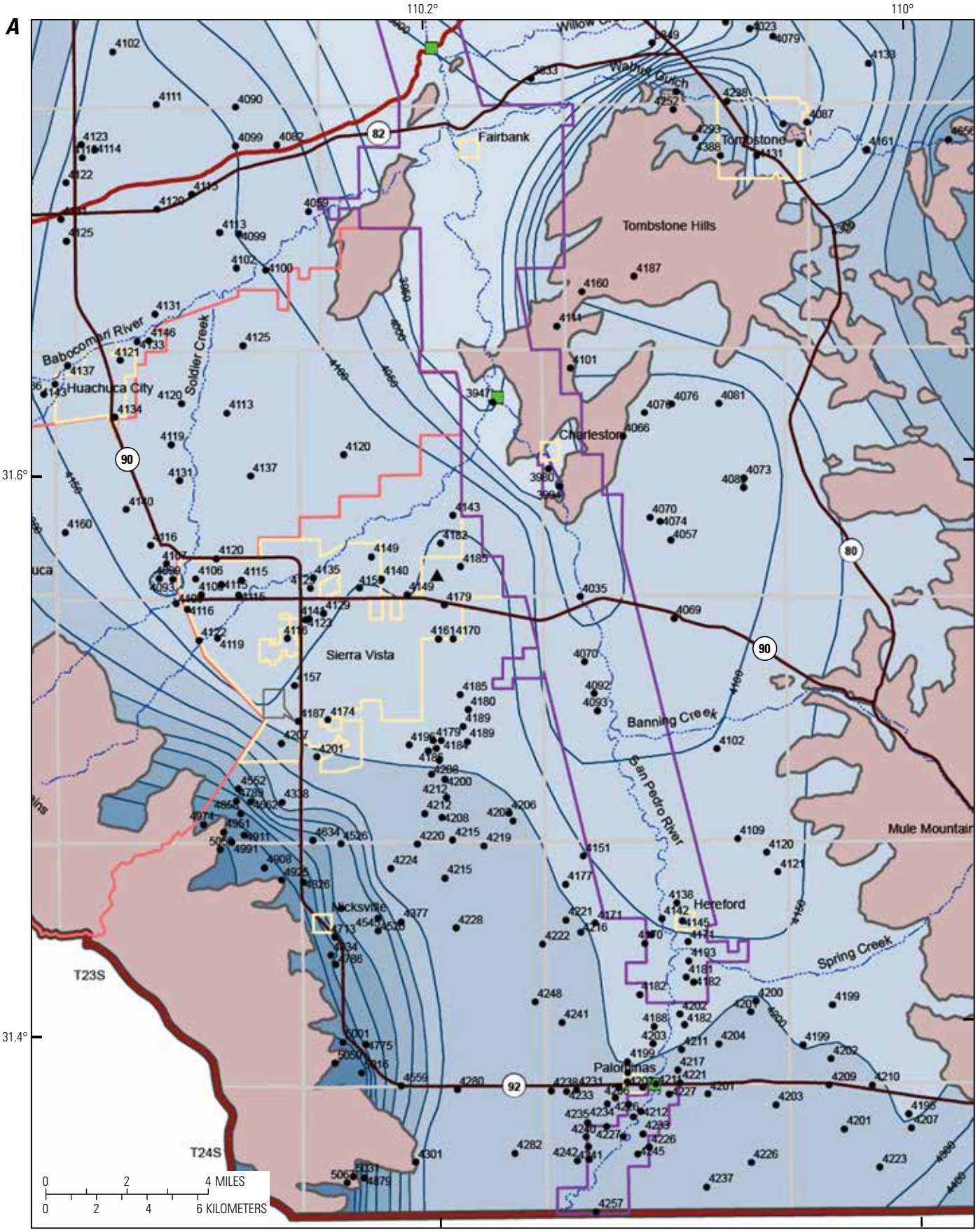


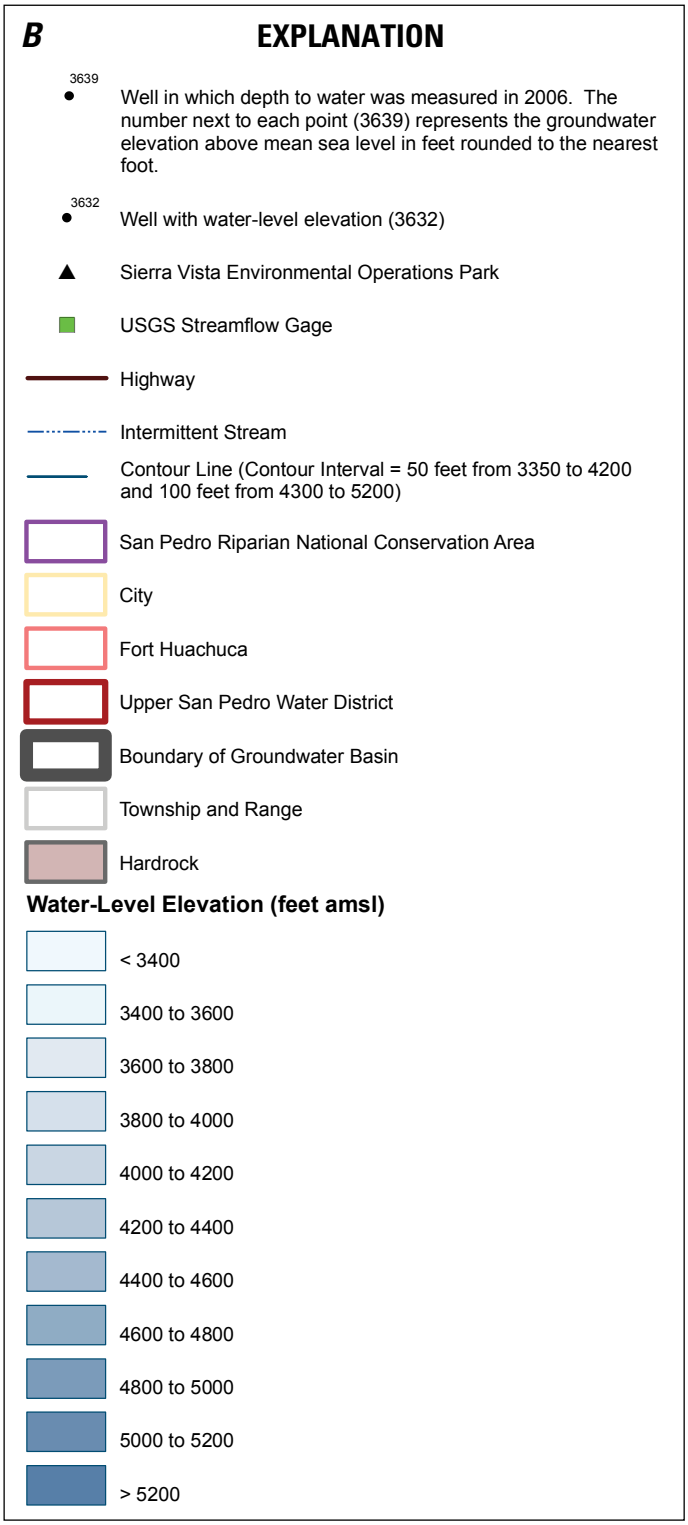
Figure 9. Graphs showing two regional-aquifer water levels (Antelope #3 and Foudy) adjacent to ephemeral streams that appear to track periods of natural recharge followed by periods of drying in the Sierra Vista Subwatershed, southeastern Arizona. Antelope #3 is south of Sierra Vista, west of the San Pedro River; Foudy is east of the San Pedro River. See figure 3 for well locations.

Overall, groundwater levels across much of the subwatershed are declining and, at least on Fort Huachuca, northeast of the municipal pumping centers, have been in steady decline since at least the 1970s (fig. 3, 4). The rate of decline is slower at wells along the eastern boundary of Fort Huachuca, furthest from the pumping centers and closest to the SPRNCA. This trend is also evident in ADWR water-level data from 2001 and 2006 that show a cone of depression centered on the pumping centers (Barnes and Putman, 2003; Schmerge and others, 2009) (fig. 10). Regional-aquifer groundwater levels in the Fort Huachuca area are clearly in decline, and this is interpreted as decreasing the tendency for groundwater to flow toward the San Pedro River.

In the region of relatively dense but unregulated lot-split development and exempt-well pumping south and east of Sierra Vista, between the Huachuca Mountains and the San Pedro River, trends were mixed with some water-level recovery, some water-level decline, and some water levels that appeared driven primarily by recharge events. East of the San Pedro River in the hills near Tombstone, Moncrief #1 was in decline in the mid-2000s when monitoring there stopped. This well was not likely affected by pumping and so the decline was most likely climate driven. In general, throughout the subwatershed, regional-aquifer water levels near to the river were declining the least and, in some cases, rising.



Modified from Schmerge and others, 2009



Modified from Schmerge and others, 2009

Figure 10. Map showing measured groundwater levels with interpolated contours across most of the Sierra Vista Subwatershed, southeastern Arizona. *A*, Map cropped and slightly modified from Schmerge and others, 2009; *B*, explanation from Schmerge and others, 2009.

Indicator 2. Horizontal Hydraulic Gradients (Regional-Aquifer Wells)—Methods

There are three well transects available to assess changes in horizontal gradients—(1) southeastern Fort Huachuca (MW3–MW4– MW5–MW1;San Pedro River), (2) central Fort Huachuca (TW1–TW8–MW7; San Pedro River), and (3) northern Fort Huachuca (TW4–TW7–TW6; Babocomari River) (fig. 11; table A5). All wells included in each transect are screened in the basin fill aquifer. Wells elsewhere in the subwatershed were not deemed satisfactory for evaluating horizontal gradients owing to poor alignment and (or) inadequate data.

Horizontal gradients are calculated as:

$$G_h = \frac{\Delta h}{\Delta l} \tag{1}$$

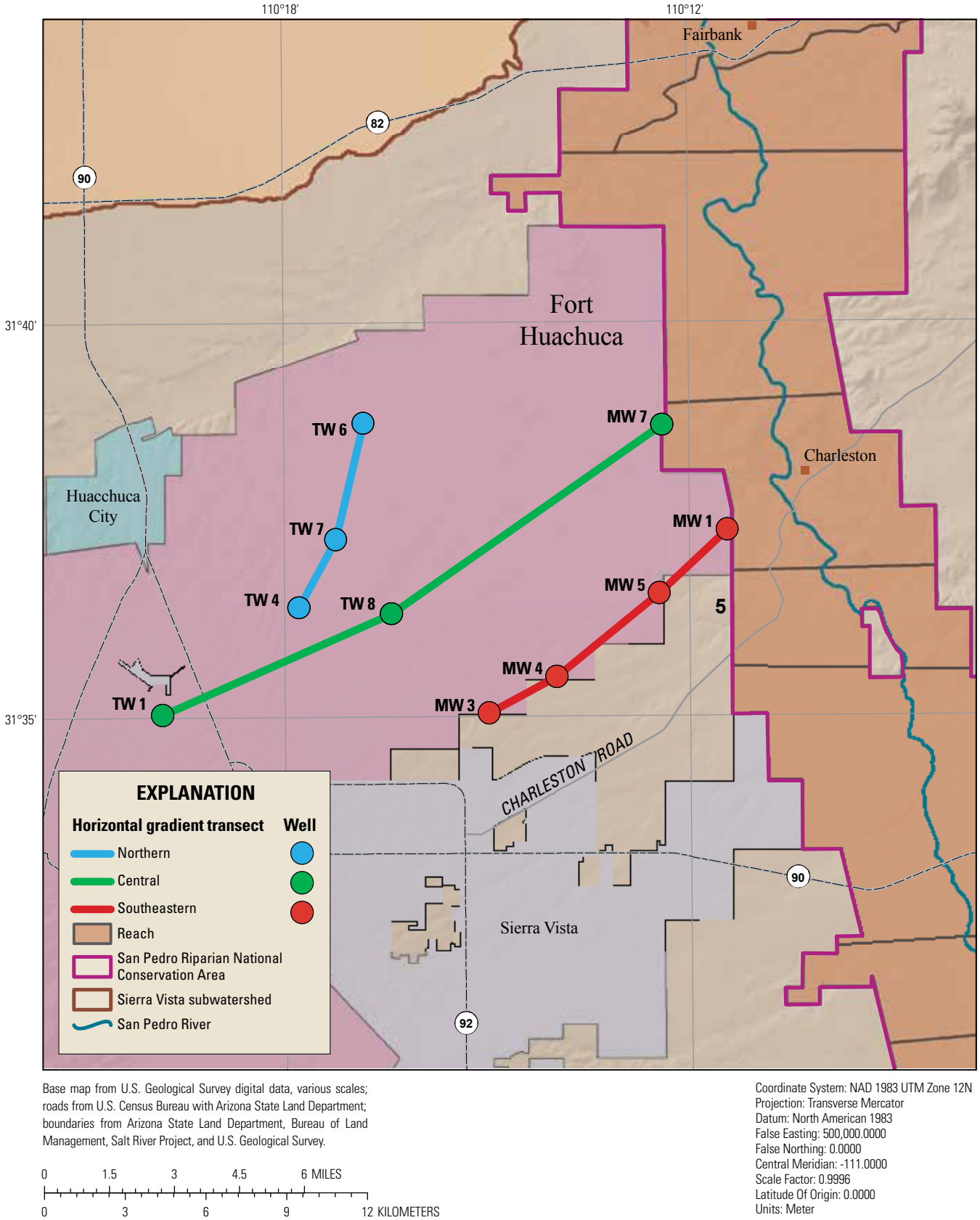
where, G_h is the horizontal gradient, Δh is the change in water level altitude from well A (nearer the pumping center) to well B (nearer the basin center), and Δl is the distance from well A to well B. Horizontal gradients have units of length per length (for example, ft per ft), and the range of horizontal gradients is typically much less than 1. A negative value indicates that the water-level elevation is lower at well A, near the pumping center, than it is at a well B, closer to the basin center. Areas with negative horizontal-gradient values would be interpreted as flowing toward the pumping center, rather than toward the riparian area and river. Areas with declining (but not necessarily negative) horizontal gradients would be interpreted as seeing a further reduction in the tendency of groundwater to flow toward the river in addition to that resulting from general reductions in groundwater levels, whereas areas of increasing horizontal gradients would be interpreted as seeing an additional increase in that groundwater flow tendency.

Indicator 2. Horizontal Hydraulic Gradients (Regional-Aquifer Wells)—Results

Horizontal gradients from the Sierra Vista-Fort Huachuca area to the San Pedro River to the northeast are in decline. This in combination with the overall reduction in Fort Huachuca groundwater levels (indicator 1) suggests that groundwater flow toward the river is decreasing. The rate of change in hydraulic gradients varies depending on the location of the well transects and the location along the transects (fig. 12*A–C*, 13*A–C*). Figure 12 shows graphically that the water-level declines over the past 20 years are greatest furthest from the river, especially for the southeastern and central transects. Water-level declines generally mean that less water can flow toward the river, but the greater reduction in water levels farther from the river further decreases the tendency for groundwater to flow toward the river.

Figure 13*A–C* plots the numerical value of the gradient as discussed in the methods section (equation 1). The most consistent decline in horizontal hydraulic gradient occurred between wells MW4 and MW5 in the southeastern Fort Huachuca transect (fig. 13*C*). Wells TW8 and MW7 in the central Fort Huachuca transect also showed a steady but somewhat smaller decline over the period of record (fig. 13*B*). These transects are roughly parallel to the groundwater gradient (perpendicular to groundwater potentiometric contour lines, such as those shown in fig. 10) and downgradient of the cone of depression (Pool and Coes, 1999; Barnes and Putman, 2003; Schmerge and others, 2006; Lacher and others, 2014). Between wells MW3 and MW4 the gradient was negative (toward the cone of depression) throughout the period of record, and continued to become more negative through about 2009, when the gradient, although still negative, stabilized (fig. 13*C*). The reason for the stabilization after years of decline is not known. The negative gradient confirms that MW3 is on the pumping side of the groundwater divide.

The horizontal gradient between wells TW7 and TW6 (northern Fort Huachuca transect; fig. 13*A*) plateaued at about 0.0009 between 1998 and 2001 and declined and then increased to 0.0008 to 0.000085, essentially plateauing again between 2003 and 2007 before declining again. Over the period of record, there was a net decrease of about 0.0001. At 0.0007 to 0.0009, these gradients are more than an order of magnitude smaller than the gradients in all but one section of the other two transects. This is probably in part because this transect is not parallel to the groundwater gradient; groundwater flow adjacent to the Babocomari River runs parallel to the river's surface flow, and this transect approaches the river at about a 45 degree angle. The gradients between wells TW1 and TW8 in the central Fort Huachuca transect (fig. 13*B*) are relatively small as well (around 0.0008) and likely owing to a similar reason—the horizontal gradient between TW1 and TW8 is close to perpendicular to the local groundwater gradient rather than parallel to it, or nearly parallel to the current groundwater potentiometric contour lines along the cone of depression.



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: -111.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Figure 11. Map showing northern, central, and southeastern horizontal well transects in the area of Fort Huachuca in Sierra Vista Subwatershed, southeastern Arizona.

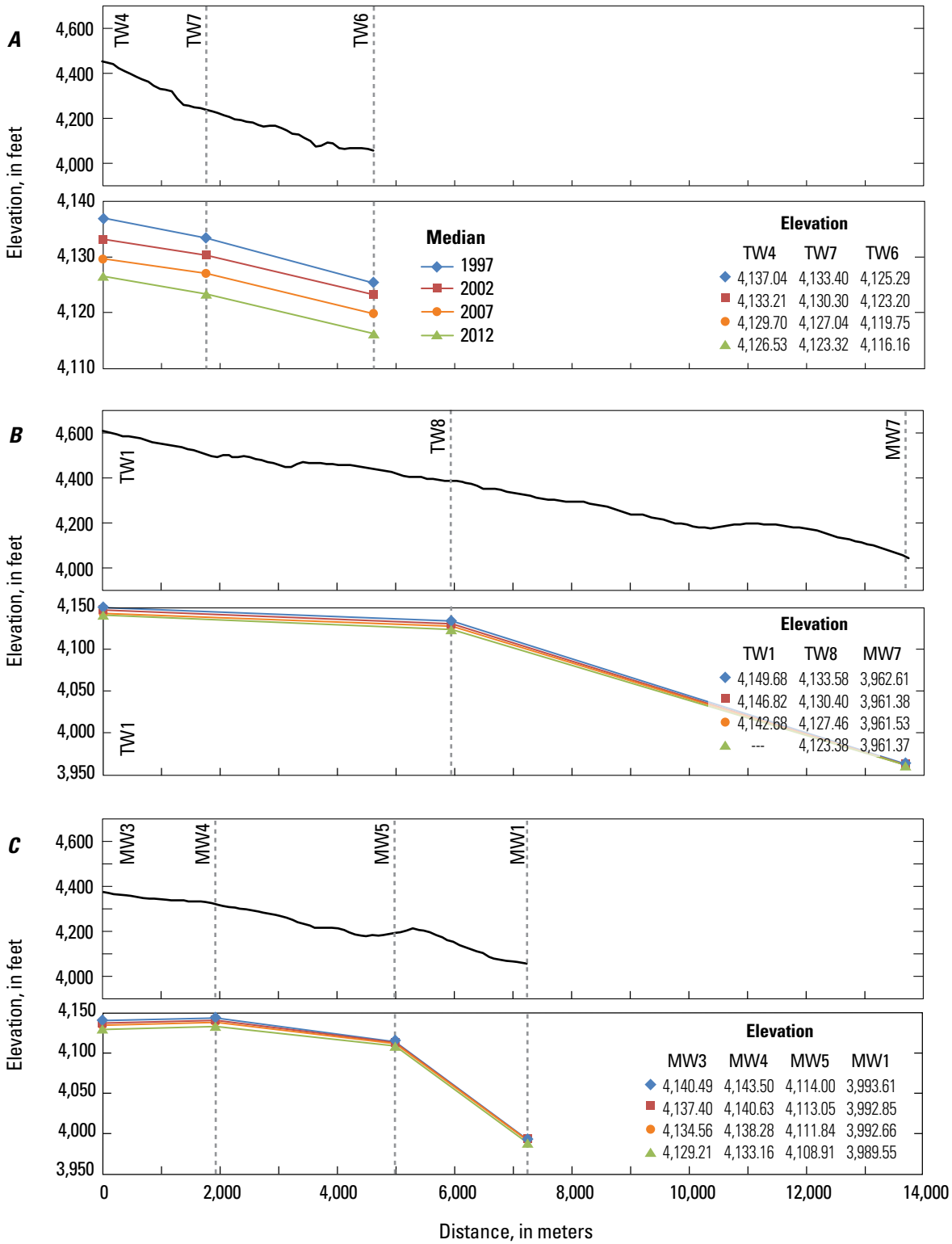


Figure 12. Graphs showing horizontal gradients expressed as annual median water levels at each well location for four separate years separated by 5-year intervals (1997, 2002, 2007, and 2012) across transects in the (*A*) northern, (*B*) central, and (*C*) southeastern parts of Fort Huachuca in the Sierra Vista Subwatershed, southeastern Arizona. In each graph, the upper plot shows land-surface elevations and horizontal separation of wells and the bottom plot shows water levels at 5-year intervals, 1997 to 2012. Scales are the same in each graph to facilitate comparison. A decreasing gradient indicates a reduction in the tendency of water to flow toward the Babocomari River (northern transect) or San Pedro River (central and southeastern transects).

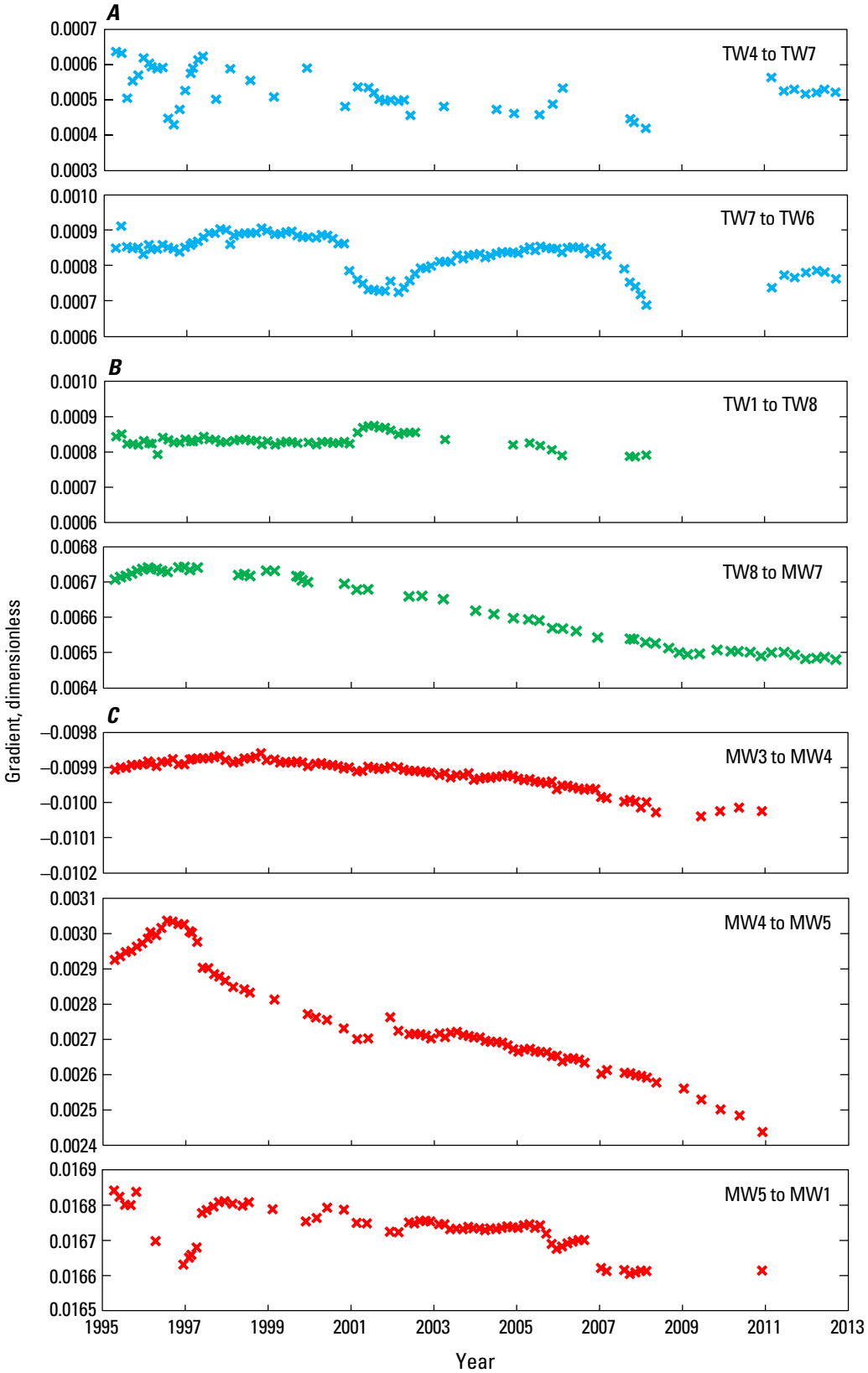


Figure 13. Graphs showing horizontal gradients at (*A*) northern, (*B*) central, and (*C*) southeastern transects across Fort Huachuca in the Sierra Vista Subwatershed, southeastern Arizona. Scales are the same in each graph to facilitate comparison. A decreasing gradient indicates a reduction in the tendency of water to flow toward the Babocomari River (northern transect) or San Pedro River (central and southeastern transects).

Indicator 3. Aquifer Storage Change Measured with Microgravity—Methods

Microgravity data are useful for measuring changes in aquifer storage and for estimating specific yield. The latter is possible when gravity-change observations are made at a well where concurrent water-level measurements are made. Then specific yield (S_y) is estimated as the slope of the best-fit straight line on a plot of change in thickness of freestanding water (determined from gravity data) versus water-level change:

$$S_y = \Delta t_{fw} / \Delta h \ , \qquad (2)$$

where Δt_{fw} is change in thickness of freestanding water measured using gravity methods and Δh is change in water level measured in a well. If the commonly used assumption that the rise in water table can be simulated as a horizontal "infinite slab" is applied, 1.06 microgal (μGal) of gravity change is equivalent to 1 inch of freestanding water.

To estimate specific yield using equation 2 requires several measurements over time—the aquifer storage change to be large enough to be measurable using gravity and the relationship between gravity and water level to be sufficiently linear (Pool, 2008). Nonlinear conditions may arise under confined aquifer conditions or where thick unsaturated zones store significant amounts of infiltrated water that has not yet reached the water table (Pool, 2008), although this does not affect the current results.

The USGS Arizona Water Science Center uses two instruments to precisely measure gravity change with time—the absolute-gravity meter measures gravity directly by timing the acceleration of a test mass in a vacuum, and the relative-gravity meter measures the difference in gravity between two stations. Relative-gravity-meter measurements must include a station where gravity does not change over time (or is assumed not to) or where gravity is measured with an absolute-gravity meter. This is similar to the way surveying instruments are used for determining elevation—relative measurements of the height difference (similar to the gravity difference) between stations are combined with known elevations (similar to absolute gravity) at one or more benchmarks (gravity stations).

To determine a single gravity value for each station, relative- and absolute-gravity measurements were combined using least-squares network adjustment. To determine regionwide gravity change from which total aquifer-storage change can be estimated, network-adjusted gravity values were differenced for successive surveys (for example, between a survey in May 2008 and one in November 2008). The differenced values were spatially interpolated by kriging.

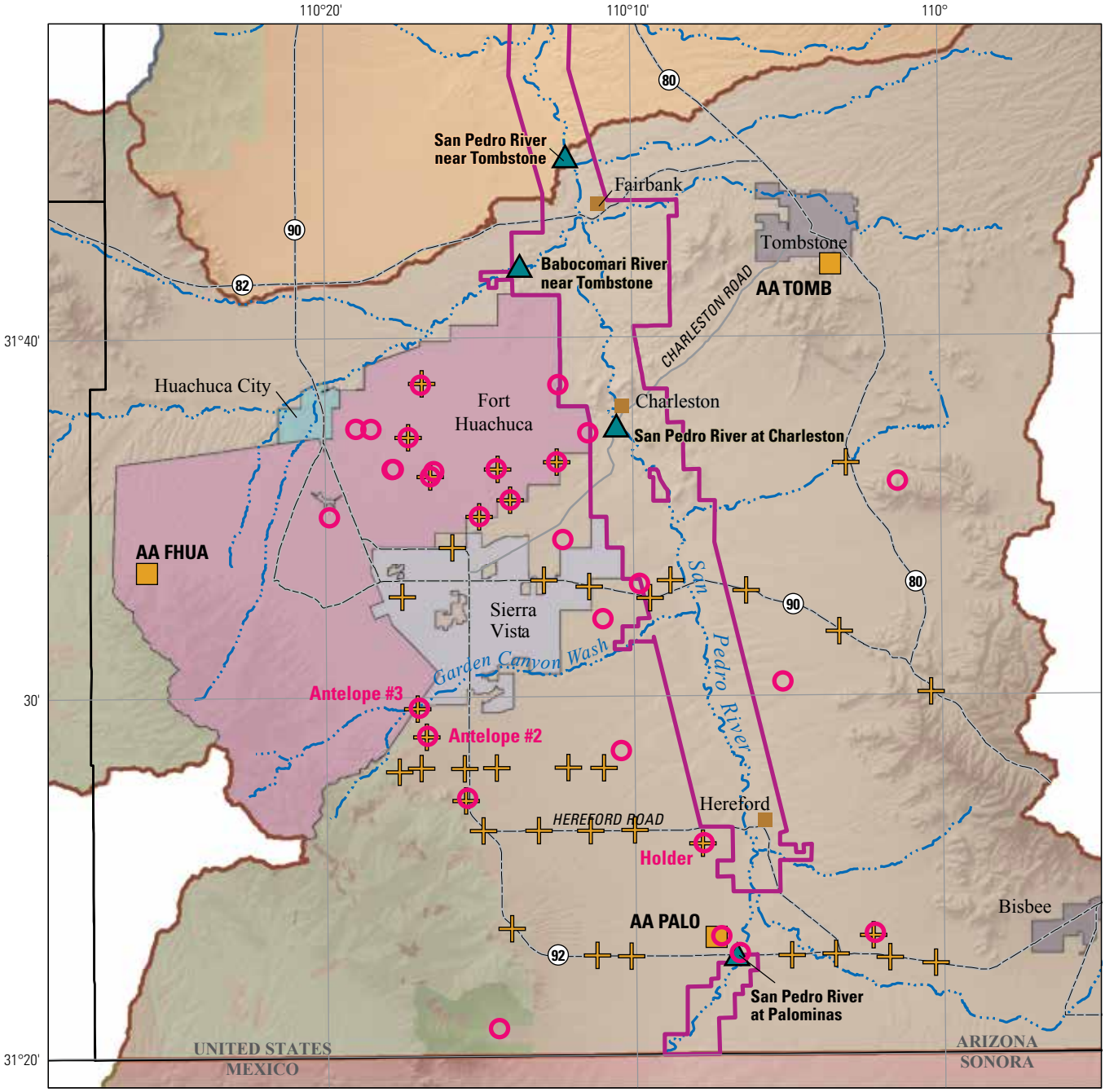
The Sierra Vista Subwatershed gravity network consists of about 45 stations (fig. 14) where gravity was measured about semiannually between 2001 and 2010 over a period of about 3 weeks. Over time, some stations were destroyed or became inaccessible, and other stations were added. During each semiannual survey, absolute-gravity measurements were made at three stations that served as reference points for observations made with a relative-gravity meter. In later years, absolute-gravity observations at six additional stations on the East Range of Fort Huachuca were made where rough access roads are problematic for the relative-gravity meter. From 2008 to 2010, additional absolute-gravity measurements were made at stations within the monitoring network to help constrain relative-gravity observations. Further details on the acquisition and processing of gravity data can be found in Kennedy and Winester (2011).

Indicator 3. Aquifer Storage Change Measured with Microgravity—Results

Gravity data indicate clear differences in aquifer storage following relatively wet summer monsoons (May 2008 to November 2008, fig. 15*A*, and May 2010 to Nov. 2010, fig. 15*E*), as compared to a dry summer monsoon (June 2009 to Nov 2009, fig. 15*C*). Following the wet summer monsoons, the November surveys showed either an increase or no change in gravity across all parts of the study area. Following the dry summer monsoon, gravity decreased across the study area. In contrast to the widespread gravity changes following either wet or dry summer monsoons, gravity change during the winter months (from November to May/June) was more focused in areas of groundwater withdrawals. In particular, the area of the pumping centers (fig. 3; northwest corner of the maps in fig. 15) showed a decrease in gravity in winter 2008, a period following a wet summer monsoon but during which there was almost no wintertime rainfall (fig. 15*B*). In contrast, an increase in gravity in the Sierra Vista area following the dry monsoon in 2009 is visible (fig. 15*D*). Wintertime rainfall in 2009–10 was greater than in any of the three prior years.

A noticeable feature on several of the gravity change maps is a "bulls-eye" increase in gravity (fig. 15*A*, *E*, *F*) at the Antelope #3 gravity station within Garden Canyon Wash (fig. 14). Although the circular pattern surrounding the station is an artifact of the interpolation process, the gravity increase at this station is real and results from stream-channel infiltration and recharge. During and following the wet monsoon in summer 2008, the water level in the Antelope #3 well in Garden Canyon Wash rose more than 50 ft (fig. 16). Little flow was recorded during the 2009 monsoon, and the water level remained stable. Following nearly continuous streamflow in winter and summer 2010, the water level rose an additional 50 ft. From a maximum in winter 2010, water levels at Antelope #3 declined steadily due to decreased streamflow in Garden Canyon Wash (fig. 16). Gravity monitoring was discontinued in 2010.

The specific yield of the aquifer in the vicinity of the gravity station in Garden Canyon Wash was estimated using the paired gravity and water level measurements at this site, using equation 2. The large variation in gravity and water levels at Antelope #3 indicated a relatively strong correlation between the two, with an R^2 value for the regression of 0.93. The calculated specific yield is 0.09 (fig. 17). The relatively rapid increase in gravity, accompanied by a rapid increase followed by a rapid decrease in water levels, indicated that unconfined aquifer conditions are present, and mounded water beneath the



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

0 1 2 4 6 8 10 MILES
0 2 4 8 12 16 20 KILOMETERS

EXPLANATION

- U.S. Geological Survey streamflow-gaging station
- Absolute gravity station
- Relative gravity station
- Basin well

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: -111.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Figure 14. Map showing gravity-station locations and regional-aquifer monitoring wells in the Sierra Vista Subwatershed, southeastern Arizona. Named stations refer to gravity-station locations mentioned in the text.

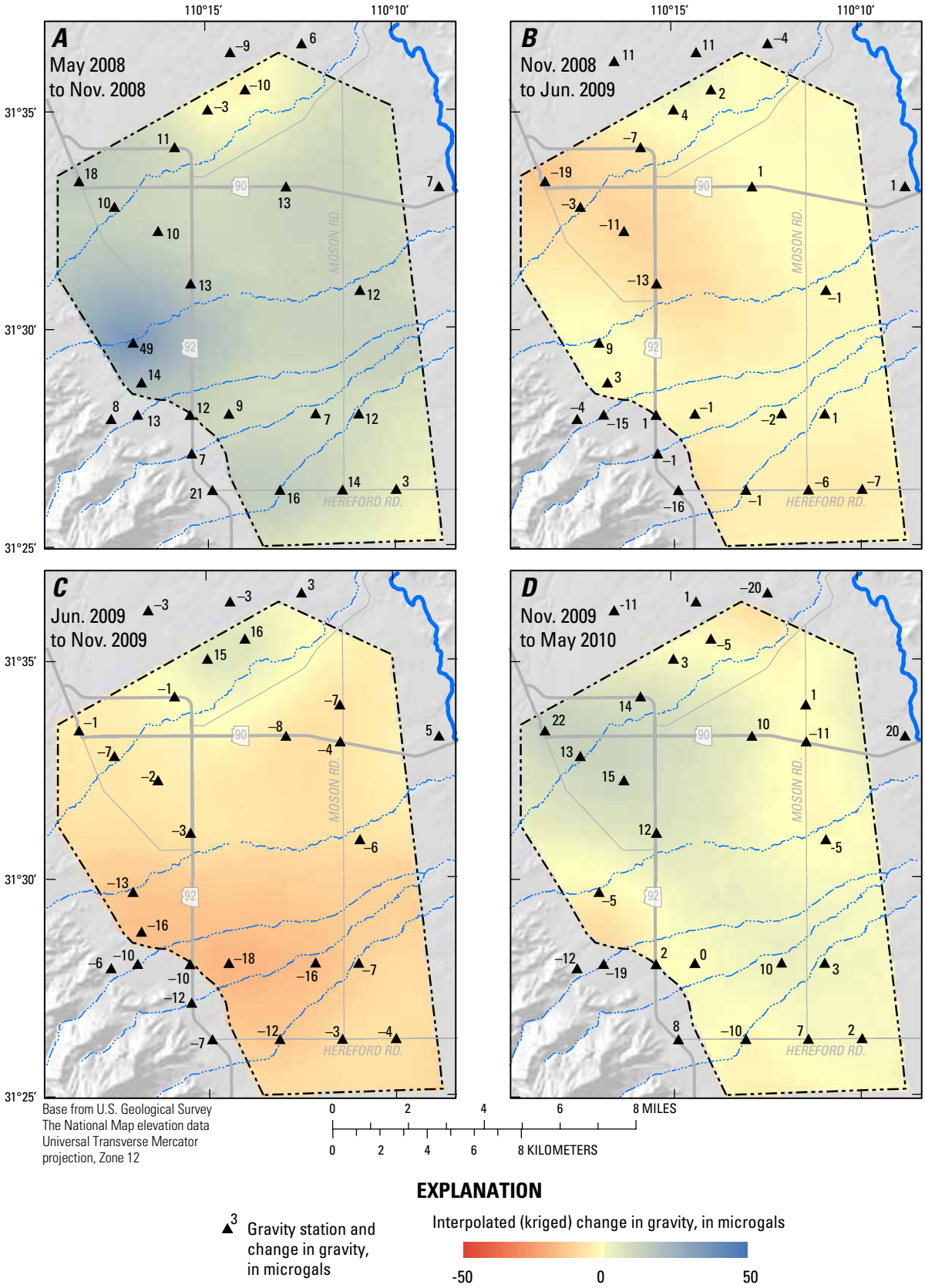


Figure 15. Maps showing gravity change in the western part of the Sierra Vista Subwatershed, southeastern Arizona, from (*A*) May to November 2008, (*B*) November 2008 to June 2009, (*C*) June to November 2009, (*D*) November 2009 to May 2010, (*E*) May to November 2010, and (*F*) May 2008 to November 2010.

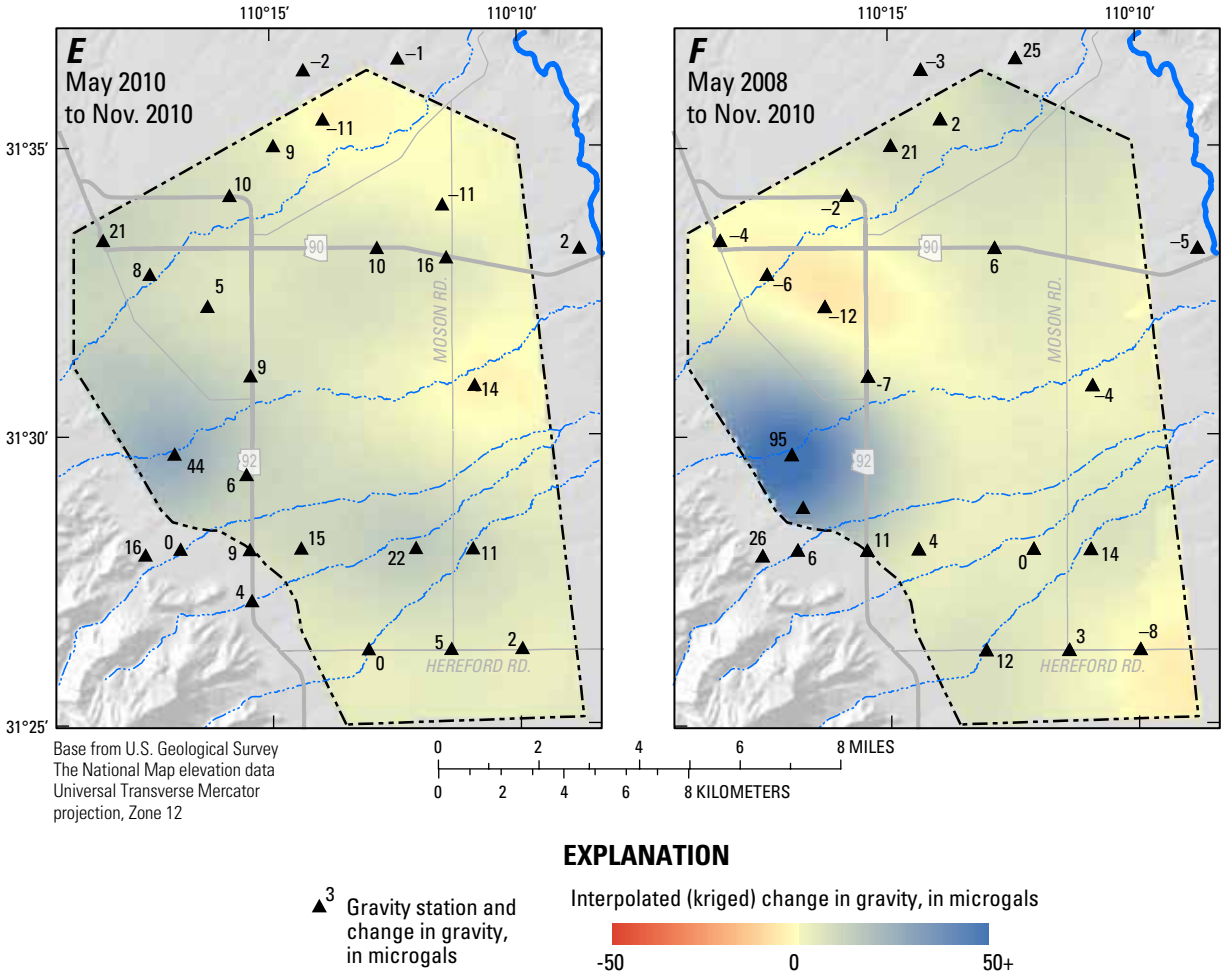


Figure 15.—Continued

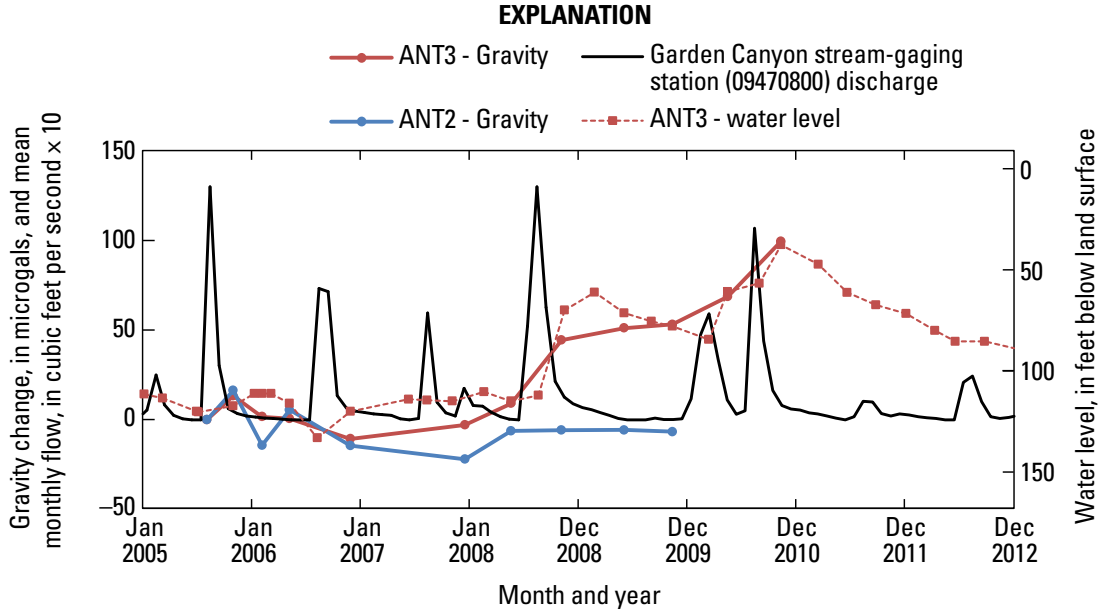


Figure 16. Graph showing water-level, streamflow, and gravity change near Garden Canyon Wash and the mountain front of the Huachuca Mountains in the Sierra Vista Subwatershed, southeastern Arizona.

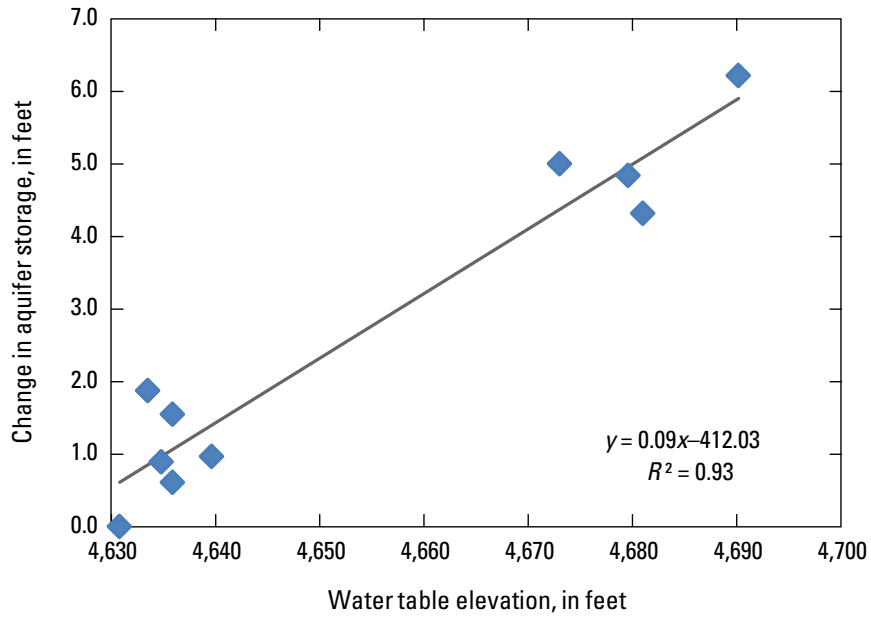


Figure 17. Graph showing specific yield determination for the Antelope #3 gravity station and well in the Sierra Vista Subwatershed, southeastern Arizona. Specific yield is equal to the slope of the trend line. R^2, coefficient of determination.

stream channel following infiltration dissipated quickly into the surrounding aquifer (removal of water through transpiration by deep-rooted mesquites is also possible).

A similar correlation of water levels with gravity is seen in the Hereford vicinity at the Holder well (fig. 18). Since the retirement of irrigated agriculture in the area in the mid-2000s, water levels have risen several feet at this site, and annual variation is much smaller than when pumping took place. The increase in aquifer storage indicated by rising water levels is also reflected in an increase in gravity from late 2006 until late 2008. Since then, however, water levels have stabilized at this well and gravity has decreased, approaching the long-term declining trend at other gravity stations in the area (fig. 18). The declining trend in gravity at station H5, −0.012 µGal/day, and at gravity station H4, −0.018 µGal/day, reflects decreasing aquifer storage. After 2008, gravity at Holder also declines (−0.033 µGal/day). The declines in gravity at H4 and H5 (all data) and Holder (post-2008 only) correspond to rates of aquifer storage decrease of 0.33±0.18 ft/yr, 0.53±0.09, and 0.94±0.12 ft/yr, respectively. The uncertainty is the standard error of the slope of the linear regression.

Indicator 4. Annual Groundwater-Budget Balance—Methods

The water-budget indicator provides a crude, single-value estimate of changes to the groundwater supply across the entire Sierra Vista Subwatershed. When the water-budget balance remains greater than 0 acre-ft, it suggests that more groundwater

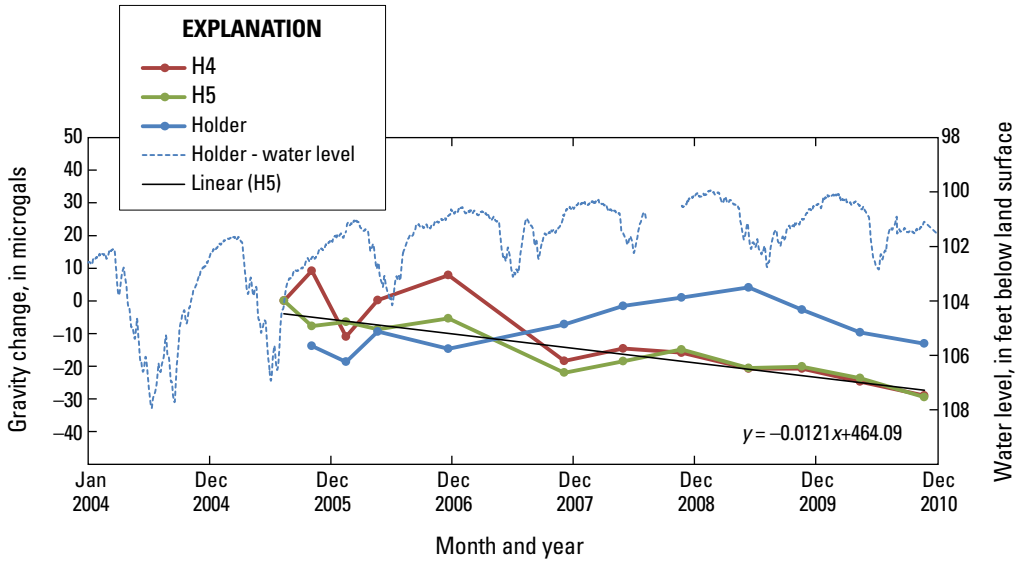


Figure 18. Graph showing water-level and gravity change at the Holder Well (312555110074301, blue curves) and gravity change at nearby stations (red and green curves) in the southern Sierra Vista Subwatershed, southeastern Arizona. Note the seasonal pumping signal in the groundwater-level data plot.

will move toward the San Pedro River. When the water-budget balance remains less than 0 acre-ft, it suggests the opposite—that less groundwater will move toward the river. When a change in the water-budget indicator might affect the San Pedro River, however, is dependent on where the primary impacts (groundwater recharge or withdrawal) are taking place. In general, it is assumed that the closer impacts are to the river, the sooner the effects will occur at the river (Leake and others, 2008).

Although the Sierra Vista and Fort Huachuca pumping center is some distance from the river (fig. 3), even if groundwater pumping were to stop today and the groundwater-budget balance was positive for decades to come, the effects of pumping over the past century will eventually capture surface flow from the river (Leake and others, 2005; Barlow and Leake, 2012). The groundwater budget is thus of little value as an annual indicator of sustainable groundwater yield. Nonetheless, it should be obvious that a subwatershed perennially in deficit will likely never see an increase in natural groundwater discharge to the river. The water-budget indicator can thus tell us when we've turned the corner toward someday seeing natural discharge to the river increasing once again, even if it is unlikely to be for decades or more.

The water budget for water years 2002–12 was calculated by determining natural and anthropogenic contributions to and withdrawals from the aquifer. Following the method used in the 321 reports (U.S. Department of the Interior, 2014), the water budget is subdivided into four categories—natural aspects of the system, groundwater pumping, active-management measures, and unintentional recharge.

Natural Aspects of the System

Natural Recharge

Natural recharge includes both recharge that occurs in ephemeral channels on a basin floor (12 to 19 percent of total natural recharge in the subwatershed; Coes and Pool, 2005) and mountain-front recharge (mountain-block recharge and recharge from intermittent and ephemeral channels at the foot of mountains). Little to no recharge is assumed to occur from direct infiltration of areal precipitation; this is considered a negligible quantity in alluvial basins with thick unsaturated zones in the arid southwest (Scott and others, 2000; Walvoord and others, 2002; Coes and Pool, 2005). Ephemeral-channel recharge includes any enhanced recharge that may occur from in-stream impoundments such as cattle tanks, although large detention basins constructed specifically to attenuate storm runoff and enhance recharge of surface flow are included in the budget as a separate element under active-management measures. The value used here for annual natural recharge is 13,500 acre-ft and represents the long-term average. This is near the middle of the range of most of the annual natural-recharge values reported elsewhere (12,520 to 15,000 acre-ft) (Freethey, 1982; Corell and others, 1996; Arizona Department of Water Resources, 2005a, b; Department of the Interior, 2005) and higher than the value used in the groundwater model of Pool and Dickinson (2007). Uncertainty for estimated annual natural recharge is about 35 percent or 4,700 acre-ft.

Groundwater Inflow

Groundwater inflow refers to the annual average volume of subsurface water that enters the subwatershed from the Sonoran Subwatershed. Groundwater outflow refers to the annual average volume of subsurface water that exits across the northern boundary of the subwatershed into the Benson Subwatershed. Freethey (1982) estimated annual groundwater inflow at 3,400 acre-ft. Corell and others (1996), U.S. Department of Interior (2004), and Arizona Department of Water Resources (2005b) all estimated inflow of 3,000 acre-ft. Pool and Dickinson (2007) simulated about 3,000 acre-ft of groundwater inflow adjacent to the San Pedro River. Mountain-front recharge from the southwestern part of the Huachuca Mountains that contributes to groundwater flow into Mexico is considered Sonoran Subwatershed recharge rather than groundwater outflow (D.R. Pool, USGS, written commun., 2014). Estimates of groundwater outflow range from 300 to 450 acre-ft, with both Corell and others (1996) and Arizona Department of Water Resources (2005b) estimating outflow of 440 acre-ft. A more comprehensive estimate totaling 1,200 acre-ft of groundwater outflow was included in the model of Pool and Dickinson (2007). This value included discharge through basin fill east (800 acre-ft) and west (100 acre-ft) of the San Pedro River, as well as discharge through the stream alluvium near the Tombstone gaging station (300 acre-ft).

Stream Base Flow

Stream base flow refers to groundwater discharged to the San Pedro River channel from the regional and (or) alluvial aquifer that then exits the subwatershed at the USGS Tombstone gaging station as surface flow (Kennedy and Gungle, 2010). Though base flow discharge can be calculated annually using the delta filter of Kennedy and Gungle (2010; see indicator 9, Base-flow on San Pedro and Babocomari Rivers), a mean annual value is more consistent with other natural water-budget values. Kennedy and Gungle (2010) provide a comprehensive survey of earlier estimates of base-flow discharge at the Tombstone gaging station. The Tombstone estimates range from 2,800 acre-feet per year (acre-ft/yr) to 7,400 acre-ft/yr, although the time periods vary. Kennedy and Gungle's (2010) average of annual base-flow discharge for the years 2002 to 2012 is about 2,600 acre-ft.

Riparian Evapotranspiration

Riparian evapotranspiration (ET) is the sum of evaporation and plant transpiration within the riparian corridors of the San Pedro and Babocomari Rivers. With the exception of a few cottonwood trees in the lower reaches of some ephemeral channels (Ramsey, Garden, Horsethief, and Moson Washes)—as well as cottonwoods and other phreatophytic plants along the now mostly perennial, effluent-dependent Curry Draw—few plants outside of the riparian corridor have roots that reach the water table (L. Levick, ARS, written commun., 2009).

Riparian ET values are taken from Scott and others (2008), using the eddy covariance technique. Previous estimates for annual subwatershed ET of the San Pedro and Babocomari Rivers combined include 7,700 acre-ft (Corell and others, 1996) and from 9,600 to 12,055 acre-ft for the year 2003 (Scott and others, 2006). Scott and others (2008) revised the 2003 values to about 13,400 acre-ft, as a result of larger estimates of groundwater use for the Babocomari River, and also provide estimates of annual ET for the other years from 2001 to 2005. The overall 2001 to 2005 5-year mean annual ET of 12,170 acre-ft is used here.

Groundwater Pumping

Municipal and Water Company Pumping

In the subwatershed, Huachuca City, Tombstone, and Fort Huachuca operate municipal water companies. Bisbee is served by a single private water company, and the Sierra Vista area and some of the other more densely developed, unincorporated parts of the subwatershed are served by more than 20 private companies. The USGS Arizona Water Science Center's Water Use (USGS AZ Water Use) program polls all water providers in the subwatershed each year and compiles the results to determine the municipal and water-company pumping volume of the water budget. This is the volume of pumped groundwater, not what is delivered to the customer, which would be less because of leakage from conveyances.

Rural/Exempt Well Pumping

Outside of water provider service boundaries, wells with pumps rated at 35 gallons per minute (56.45 acre-ft/yr) or less are generally exempt, by State law, from regulation, including any metering or water-use fees. Such rural/exempt well pumping is difficult to estimate. The rural/exempt well pumping water-budget value used here is based on the average of the ranges reported in a recent analysis of exempt well pumping in the subwatershed (Plateau Resources LLC, 2013). Plateau Resources LLC (2013) determined an average use per exempt well of 0.24 acre-ft/well. To account for the reduced water conservation that comes with unmetered water service (Hanke and Flack, 1968; Gallagher and Robinson, 1977; Hanke and de Mare, 1982; Litke and Kauffman, 1993; Walters and Young, 1994; Stedman, 2006; Godley and others, 2008; Environment Canada, 2011; Arizona Department of Water Resources, 2010), the residential part of Plateau Resources LLC (2013) exempt well-pumping estimates were up-scaled by a factor of 5 percent.

Rural/exempt well pumping values for the subwatershed from 2002 to 2011 were back-calculated using the Plateau Resources LLC (2013) 2012 average use per well value (0.24 acre-ft/well) in the equation:

$$R_{i=2002–2011} = R_{2012} + 1.05 \times (0.24 \text{ acre-ft/w} \times W_{i+1}), \qquad (3)$$

where R is rural/exempt well pumping in acre-ft, W is the number of new rural/exempt well households in year (i+1), and R_{2012}= −1,445 acre-ft (Plateau Resources LLC, 2013, upscaled by a factor of 0.05).

Industrial Pumping

Industrial pumping includes turf irrigation, sand and gravel mining, and stock-tank pumping for cattle not served by a water provider.

Turf Irrigation

Turf irrigation data are collected by the USGS AZ Water Use program, and comes from pumping data from Turquoise Valley Golf Course near Naco, Pueblo del Sol Country Club in Sierra Vista, and City of Sierra Vista pumping to water turf at Veteran's Memorial Park in Sierra Vista.

Sand and Gravel Operations

Water used by subwatershed sand-and-gravel operations was calculated using the method of Arizona Department of Water Resources (2005b) by multiplying the annual increase in population by 0.201 acre-ft/person. This relation is derived from 1985 to 1990 subwatershed water-use figures and related population data (Arizona Department of Water Resources, 2005b). In years when the population change is negative (for example, 2007), water use by sand-and-gravel operations is estimated to be zero. Population data for 2002 to 2012 are from the State of Arizona's Office of Employment and Population Statistics (Arizona Department of Administration, 2013) with the exception of 2010 data, which are from the U.S. Census Bureau (Arizona Department of Administration, 2013).

Stock-Tank Pumping

Stock-tank pumping is not metered. An accurate estimate depends on a good head count of cattle. For 2012, Hereford Natural Resource Conservation District (NRCD) volunteers estimated 3,200 animal units in the subwatershed. Each animal unit is estimated to consume 20 gallons of water per day (about 0.0224 acre-ft/yr) per animal, on average, and that 80 percent of the water consumed is groundwater. The number of cattle in the subwatershed has been static since 2008; previous to that, numbers were declining about 10 percent per year beginning in 1987 (James Lindsey, Chairman, Hereford NRCD, written commun., 2013).

Irrigation Pumping

The USGS AZ Water Use program has estimated the consumptive use of subwatershed irrigation pumping since 2007. In 2012, they estimated 55 acre-ft of pumped groundwater was consumptively used by crops. The USGS AZ Water Use program has used the ADWR groundwater-basin delineation for the subwatershed boundary over the past decade and that delineation is retained here. This estimate excludes much of the pumping for vineyard, orchard, and pasture irrigation that occurs in this area.

Active-Management Measures

Mesquite and Tamarisk Treatment

The effective impact of mesquite (*Prosopis velutina*) and tamarisk (*Tamarix spp.*) treatment on aquifer storage is derived from the work of Scott and others (2008) and Stromberg and others (2009); an acre of velvet mesquite in the Sierra Vista Subwatershed with 100-percent canopy uses about 2.27 acre-ft/yr of water on average. Similar work has not been conducted specific to tamarisk in the subwatershed, but given similarities in shrub size and effective surface area of the canopy, it is assumed that the water use of tamarisk is similar to that of mesquite (R.L. Scott, Agricultural Research Service, written commun., 2013).

The reduction of groundwater use from a given mesquite eradication treatment is assumed to be about 10 percent per year (Uchytil, 1990; Huang and others, 2007). At 10 years post treatment, no further effect is assumed. BLM reported treatments of varying scope in 2002, 2008, 2009, 2010, and 2011 and estimated a canopy cover of about 15 percent in 2008, which is assumed here for all treatment areas (N. Dietrich, BLM, written commun., 2008, 2009; M. Radke, BLM, written commun., 2011). Long-term recovery of treated tamarisk is assumed to be similar to that of mesquite (R.L. Scott, Agricultural Research Service, written commun., 2013).

Municipal-Effluent Recharge

Sierra Vista and Fort Huachuca recharge treated effluent in detention ponds adjacent to their individual wastewater reclamation facilities and calculate pond infiltration, which is assumed to recharge the aquifer beneath the ponds at each facility. The Sierra Vista Public Works Department also reports leakage through their constructed wetlands at their Environmental Operations Park (EOP). Estimated annual leakage varied by year from 2004 to 2012 from a high of 800 acre-ft in 2008 to a low of 350 acre-ft in 2009 and 2010. In 2011 and 2012, leakage was directly estimated from influent, effluent, precipitation, evaporation, and assumed ET (Dooley, 2013). Tombstone and Bisbee both discharge treated effluent into ephemeral channels (Walnut Gulch and Greenbush Draw, respectively), a percentage of which is assumed to recharge. The sum of these four treated effluent recharge values comprises the total municipal effluent recharge water budget item. Length of surface flow from the Tombstone wastewater treatment facility is about 0.25 mi. Surface flow in Greenbush Draw that originates from the Bisbee treatment facility discharge pipe is estimated at 1.3 mi (Russ McConnell, Bisbee Public Works Department, written commun., 2009), although satellite images showed as much as 2 mi or more of surface flow. Tombstone annual effluent discharge volumes from 2008 to 2012 were obtained from the Tombstone Public Works Department. Tombstone discharge values for other years were calculated using the average per capita effluent discharge for 2008 and 2009 (62 gallons per capita per day) (gpcd). This value was then scaled by population for years 2002 to 2007. All Bisbee data were obtained from the Bisbee Public Works Department except for 2007, when data were not available and the previous year's effluent discharge was used as the estimated volume. Tombstone and Bisbee total effluent discharge values are reduced by 4 percent to account for ET. This is based on recharge estimates for the Santa Cruz River in Pima County, adjusted for the less dense vegetative canopy in Greenbush Draw and Walnut Gulch (Leenhouts and others, 2006; Nathan Lehman, Bureau of Reclamation, written commun., 2014).

Detention-Basin Recharge

Detention-basin recharge was calculated for five detention basins on Fort Huachuca (Greely Catch, Graveyard Gulch, Runway, Soldier, and Stormwater One) by the USGS and for five detention basins in Sierra Vista (Busby, Country Club, Summit, 7th Street, and Rostron) by ARS. The USGS and ARS both calculate the detained water volume (from bathymetric surveys in concert with basin stage) integrated over time minus discharge volume from the basin (if any) and minus evaporation that occurs while water is pooled. Beyond this, calculation methods diverge somewhat but are considered comparable (S. Tadayon, USGS, written commun., 2010; T. Keefer, ARS, written commun., 2010, 2013).

Unintentional Recharge

Incidental Recharge

Incidental recharge includes water that recharges the aquifer as a byproduct of water-use activities not specifically designed to recharge the. In the subwatershed this includes septic systems, leach fields, turf facilities such as golf courses and parks, and water supply systems. Lehrner (1990) observes that all water supply systems leak and assumes that 10 to 50 percent of the source water can be lost from systems.

Calculated as the difference between water pumped and water delivered to water meters, about 400 acre-ft of water was lost from subwatershed municipal water systems in 2012. The loss likely occurred as a mix of actual leaks, under-recording delivery meters, unmetered water taken from hydrants, and perhaps water theft (Arizona Department of Water Resources, 2005b). The actual leaks in the system would be relatively near the surface, probably within the root zone, and thus much of the leaked water could be expected to evapotranspire rather than recharge. Some water, particularly from larger leaks, could infiltrate to the water table, but because this amount is highly uncertain, no incidental recharge from water-system leakage is incorporated into the water budget.

Koehler (2000) estimated incidental recharge of discharge to septic fields (from interior use) with a tile field less than 5 ft deep to be about 40 percent of discharge. Pool and others (2011) estimated that 35 percent of the total water withdrawn from an exempt well recharges the aquifer, or about 50 percent of total interior use based on Plateau Resources LLC (2013) estimate of interior use (about 70 percent of total water pumped).

Septic-system recharge is calculated here as the total population on septic systems (a combination of the exempt well population, the unincorporated population on water-company water but not on sewer, and the incorporated population who have septic systems) times 50 percent times the estimated average interior use per person (0.067 acre-ft per person per year or about 25 gpcd; Plateau Resources LLC, 2013). This assumes that some fraction of septic system tile fields in the subwatershed are deeper than 5 ft. Incidental recharge from golf courses and other turf irrigation is estimated as 5 percent of total water spread on the turf (Arizona Department of Water Resources, 2005b).

Urban Enhanced Recharge

Urban enhanced recharge, also referred to as "focused urban runoff recharge" in the literature (for example, Gallo and others, 2013), is runoff that is concentrated from rooftops, pavement, and compacted soils. In temperate to tropical climates, diffuse or aerially distributed recharge is common. In arid or semiarid climates, where evaporation greatly exceeds precipitation, little diffuse recharge can occur. Instead, recharge takes place where surface runoff is concentrated—along mountain fronts and in depressions and ephemeral stream channels (Eastoe and others, 2004; Pool, 2005; Coes and Pool, 2005; Scanlon and others, 2006; Carlson and others, 2011). Impervious and semi-impervious surfaces common in urbanized areas typically increase the volume of concentrated surface runoff (Goodrich and others, 2004; Carlson and others, 2011; Gallo and others, 2013; Kennedy and others, 2013).

Urban enhanced recharge in the subwatershed was estimated using National Land Cover Database (NLCD) low, medium, and high-intensity developed land-cover classes. Estimates of enhanced recharge attributed to urbanization in the Coyote Wash Watershed in and east of the City of Sierra Vista (GeoSystems Analysis, 2004) were scaled up to the entire subwatershed based on the ratios of NLCD developed land cover classes in the subwatershed (derived from 2001, 2006, and 2011 NLCD imagery) to those in the Coyote Wash Watershed for the same years (Lainie Levick, ARS, written commun., 2015). The estimates were about 1,782 acre-ft for 2001, 2,135 acre-ft for 2006, and 2,330 acre-ft for 2011. Increases in developed land cover for other years were estimated based on population change relative to the 2001, 2006, and 2011 land covers.

Uncertainty

The annual water-budget uncertainty is an attempt to numerically quantify our confidence in the water-budget balance and each of the water-budget elements. Uncertainty in this case is the statistical error in the calculated values and is quantified as the calculated or estimated standard deviation. Uncertainty is distinct from variability, which in most cases can be explained by natural climate fluctuations (Lehmann and Rillig, 2014). The annual water-budget uncertainty was calculated using a randomization scheme that was carried out by generating random values from assumed probability distributions for each budget component. The uncertainty in the subwatershed annual water budget is about 5,500 acre-ft (table 2).

Indicator 4. Annual Groundwater-Budget Balance—Results

The estimated annual water budget was in deficit in 2002 (−10,700±5,500 acre-ft) and remains in deficit in 2012 (−5,000±5,500 acre-ft), but the annual deficit has decreased by about 5,700 acre-ft. Much of this decrease in the deficit can be traced to reduced agricultural irrigation pumping (about 2,400 acre-ft), reduced municipal pumping (about 1,200 acre-ft, of which reductions on Fort Huachuca account for nearly one-half since 2002), and enhanced recharge of treated effluent (about 1,700 acre-ft). Unintentional recharge accounts for about 600 acre-ft of additional recharge from 2002 to 2012, although the uncertainty of this component is large.

Annual precipitation appears related to pumping in many years. Precipitation was below the 11-year mean of 14.61 in the subwatershed in 2002, 2003, 2009, 2011, and 2012 (fig. 2, table 3). Municipal pumping was up in most of those years as a result, especially in 2009 and less so in 2011. These changes appeared to help drive the increases in anthropogenic effects on aquifer storage in 2009 and 2011 (table 3 and fig. 19). By 2012, municipal pumping dropped again. Although municipal pumping peaked in 2005 at about 11,100 acre-ft, more than 1,000 acre-ft of irrigation pumping was retired that year, resulting in the initiation of a trend in annual reductions in subwatershed groundwater pumping that, with the exception of 2009, continued through 2012 despite ongoing population growth (table 3).

The uncertainty in the groundwater budget results from the lack of accurate data for a number of systems, including natural recharge to and discharge from the subwatershed aquifer, and some types of groundwater pumping. The absolute annual value of the water-budget balance is thus not well suited to use as a primary indicator of sustainability. The best use of the water budget as an indicator is when the focus is on the change in the water-budget balance that is attributed to human activities (table 3 and fig. 19). For those water-budget components controlled by human use and management, from 2002 to 2012 there was a reduction in the total amount of annual aquifer storage loss of about 5,100 acre-ft. Nearly three-quarters of the reduced storage loss can be attributed to an overall reduction in groundwater pumping in the subwatershed during that period. The rest of the reduced storage loss was primarily because of increasing recharge of treated effluent from Sierra Vista, Fort Huachuca, Tombstone, and Bisbee. However, essentially all of this change occurred from 2002 to 2007. Total annual effect on storage due to human activities remained between −8,500 acre-ft and −9,100 acre-ft from 2008 to 2012. As discussed in the paragraphs above, the total aquifer-storage deficit is less than this due to the effects of unintentional recharge and a surplus of natural recharge over natural discharge—groundwater that otherwise would discharge naturally is instead intercepted by pumping.

Table 2. Groundwater-budget values calculated for 2002 and 2012, change in values, and uncertainty for natural aspects of the system, groundwater pumping, active-management measures, and unintentional recharge in the Sierra Vista Subwatershed, southeastern Arizona.

[acre-ft, acre-foot; NA, not applicable]

Groundwater budget element, in acre-ft	2002	2012	Change	Estimated uncertainty
1. Natural recharge	13,500	13,500	NA	±4,700
2. Groundwater inflow	3,000	3,000	NA	±800
3. Groundwater outflow	−1,200	−1,200	NA	±400
4. Stream base flow discharge	−2,600	−2,600	NA	±1,100
5. Riparian evapotranspiration	−12,200	−12,200	NA	±900
Subtotal: Natural aspects of system	500	500	NA	NA
6. Municipal and water-company pumping	−10,700	−9,500	1,200	±1,000
7. Rural/exempt well pumping	−1,200	−1,400	−200	±700
8. Industrial pumping	−1,600	−1,200	400	±300
9. Irrigation pumping	−2500	−50	2,450	±100
Subtotal: Groundwater pumping	−16,000	−12,150	3,850	NA
10. Effective impact of mesquite and tamarisk treatment	400	100	−300	±70
11. Municipal-effluent recharge	1,300	3,000	1,700	±200
12. Detention basin recharge	200	100	−100	±100
Subtotal: Active-management measures	1,900	3,200	1,300	NA
13. Total incidental recharge	900	1,000	100	±700
14. Urban enhanced recharge	1,900	2,400	500	±1,000
Subtotal: Unintentional recharge	2,800	3,400	600	NA
Total water-budget balance	−10,800	−5,000	5,800	±5,500
Sierra Vista Subwatershed population	69,942	80,866	10,924	NA

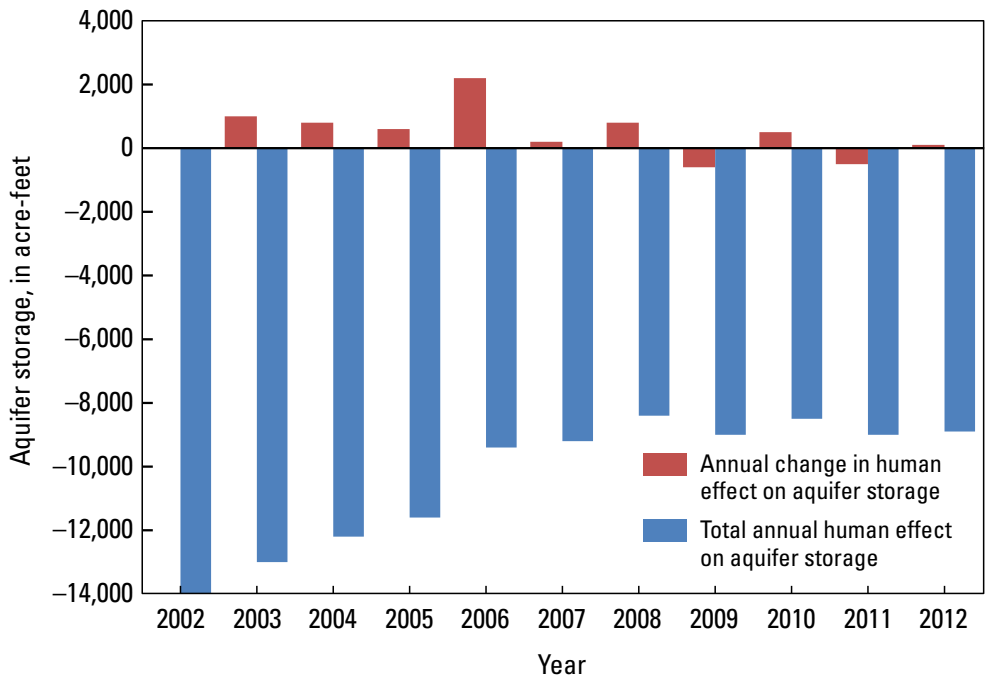


Figure 19. Bar graph showing total annual human effect on the water-budget balance and the year-to-year change in that total annual human effect from 2002 to 2012 in the Sierra Vista Subwatershed, southeastern Arizona.

Table 3. Annual values, mean value, change in value, and uncertainty calculated for water-budget elements that are directly attributable to human water use and water-management actions in the Sierra Vista Subwatershed, southeastern Arizona.

[acre-ft, acre-foot; NA, not applicable]

Water-budget element, in acre-ft	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	Mean	Total change	Uncert-ainty
Municipal and water-company pumping	−10,700	−10,900	−10,600	−11,100	−10,600	−10,600	−9,900	−10,100	−9,800	−9,800	−9,500	−10,300	1,200	1,000
Rural/exempt well pumping	−1,200	−1,200	−1,300	−1,300	−1,400	−1,400	−1,400	−1,400	−1,400	−1,400	−1,400	−1,300	−200	700
Industrial pumping	−1,600	−1,300	−1,700	−1,400	−1,300	−900	−1,100	−1,100	−1,300	−1,200	−1,200	−1,300	400	300
Irrigation pumping	−2,500	−2,500	−2,500	−1,500	−400	−300	−300	−300	−50	−50	−50	−950	2,450	100
Subtotal: Groundwater pumping	−16,000	−16,000	−16,000	−15,300	−13,700	−13,100	−12,700	−12,900	−12,700	−12,600	−12,200	−13,900	3,800	NA
Effective impact of mesquite (*Prosopis velutina*) and tamarisk (*Tamarix* spp.) treatment	400	400	300	300	200	200	300	200	200	100	100	200	−300	70
Municipal effluent recharge	1,300	2,300	3,100	3,200	3,600	3,400	3,700	3,500	3,500	3,300	3,000	3,100	1,700	200
Detention basin recharge	200	200	300	100	300	300	200	50	400	100	100	2,000	−100	100
Subtotal: Active management measures	1,900	2,900	3,700	3,600	4,100	3,900	4,200	3,800	4,100	3,500	3,200	3,500	1,300	NA
Total annual human effect on aquifer storage	−14,100	−13,100	−12,300	−11,700	−9,600	−9,200	−8,500	−9,100	−8,600	−9,100	−9,000	−10,400	5,100	1,300
Annual change in human effect on aquifer storage	NA	1,000	800	600	2,100	400	700	−600	500	−500	100	510	NA	NA
Precipitation (4 station mean; inches)	11.60	12.71	15.40	14.75	16.12	17.26	17.65	10.37	20.52	10.91	13.45	14.61	NA	NA
Sierra Vista Subwatershed population:	69,942	70,343	72,911	74,345	75,774	75,703	76,723	77,106	78,884	80,048	80,866	75,695	10,924	NA

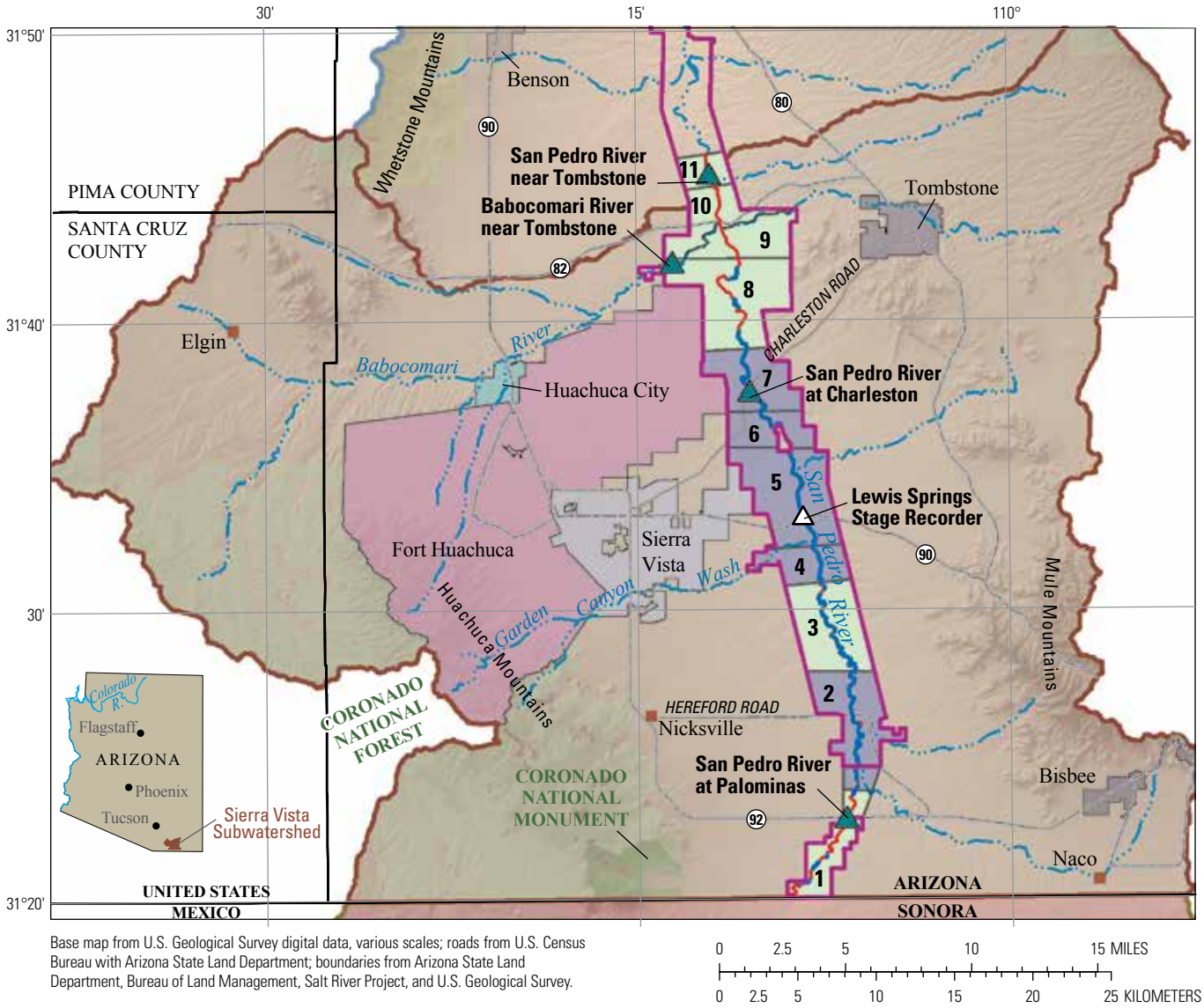
Even if groundwater pumping were to stop today and the groundwater budget balance was positive for decades to come, the effects of pumping over the past century would eventually capture surface flow from the river (Leake and others, 2005; Barlow and Leake, 2012). According to recent modeling, some capture of surface flow from the San Pedro River is already occurring (Lacher and others, 2014), although widespread capture was not yet obvious based on many of the indicators and related monitoring found in this report.

Group 2. Riparian-System Indicators

Indicator 5. Near-Stream Alluvial-Aquifer Groundwater Levels—Methods

Alluvial groundwater (or bank storage) can come from flood flows and enhanced-infiltration projects, as well as from a regional aquifer. Regardless of the source, increases in alluvial-groundwater levels are a direct and immediate indication of an increase in groundwater available to riparian plants and to discharge to the San Pedro River; decreases in alluvial-groundwater levels conversely indicate a loss of available groundwater for plants and the river. Stromberg and others (2006) found that the lowest dry-season water levels are well correlated with riparian-system health.

Near-stream alluvial-aquifer groundwater levels were measured in monitoring wells (nonpumping) screened in the alluvial aquifer adjacent to the San Pedro River and include wells screened in either the post- or pre-entrenchment alluvium (Pool and Coes, 1999; Leenhouts and others, 2006; table A5). Alluvial-groundwater-level data were analyzed for 10 of the 11 subwatershed river reaches identified by Stromberg and others (2006) (fig. 20). Water levels were measured on an approximately quarterly basis. All wells except Moson (MOSLNS, reach 6) were outfitted with continuously recording pressure transducers (measurement intervals ranged from 0.5 to 12 hours), although all wells had periods of varying length with missing continuous data (fig. 21). A single well, representative of the local alluvial-groundwater levels and with suitable water-level data, was selected for analysis in each reach (fig. 22). When possible, a well situated on the post-entrenchment flood plain, typically within the cottonwood-willow forest, was used to evaluate conditions representative of the riparian forest in that reach. For reaches



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

EXPLANATION

- Intermediate - class 2
- Wet - class 3
- Benson Subwatershed
- Sierra Vista Subwatershed
- Sonoran Subwatershed
- 1 Reach number

San Pedro River, June 2012

- Dry
- Wet
- U.S. Geological Survey streamflow-gaging station
- DCP stage recorder gage
- Reach
- San Pedro Riparian National Conservation Area

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: –111.0000
Scale Factor: 0.9996
Latitude of Origin: 0.0000
Units: Meter

Reach Numbers and Names

1. Palominas	4. Cottonwood	7. Charleston	10. Fairbank North
2. Hereford	5. Lewis Springs	8. Charleston Mesquite	11. Tombstone
3. Hunter	6. Moson	9. Fairbank South	

Figure 20. Map showing riparian ecological condition classes for the 11 reaches in the Sierra Vista Subwatershed, southeastern Arizona, identified in Leenhouts and others (2006) (modified from Stromberg and others, 2006).

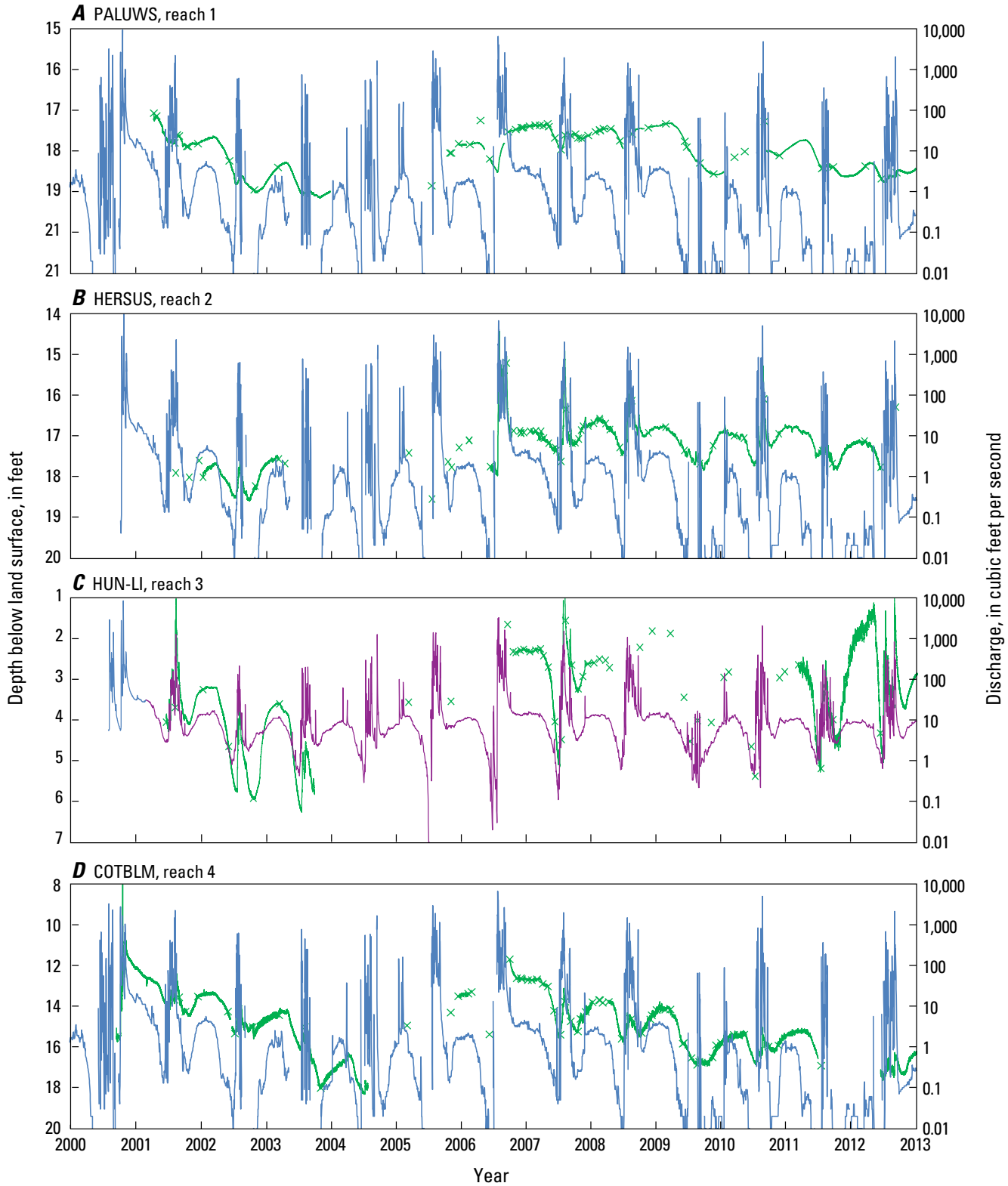


Figure 21. Graphs showing groundwater-table elevation for alluvial aquifer wells plotted with stream discharge from the nearest San Pedro River gaging station in the Sierra Vista Subwatershed, southeastern Arizona. *A*, PALUWS, reach 1 (Palominas); *B*, HERSUS, reach 2 (Hereford); *C*, HUN-LI, reach 3 (Hunter); *D*, COTBLM, reach 4 (Cottonwood); *E*, LSP-1, reach 5 (Lewis Springs); *F*, CHB-LI, reach 7 (Charleston); *G*, FBK-LI, reach 10 (Fairbank); *H*, TOM-LI, reach 11 (Tombstone). In *E*, a detailed insert of the record at LSP-1, October, 2009 to October, 2010, shows changes in water levels likely related to beaver damming of the stream channel.

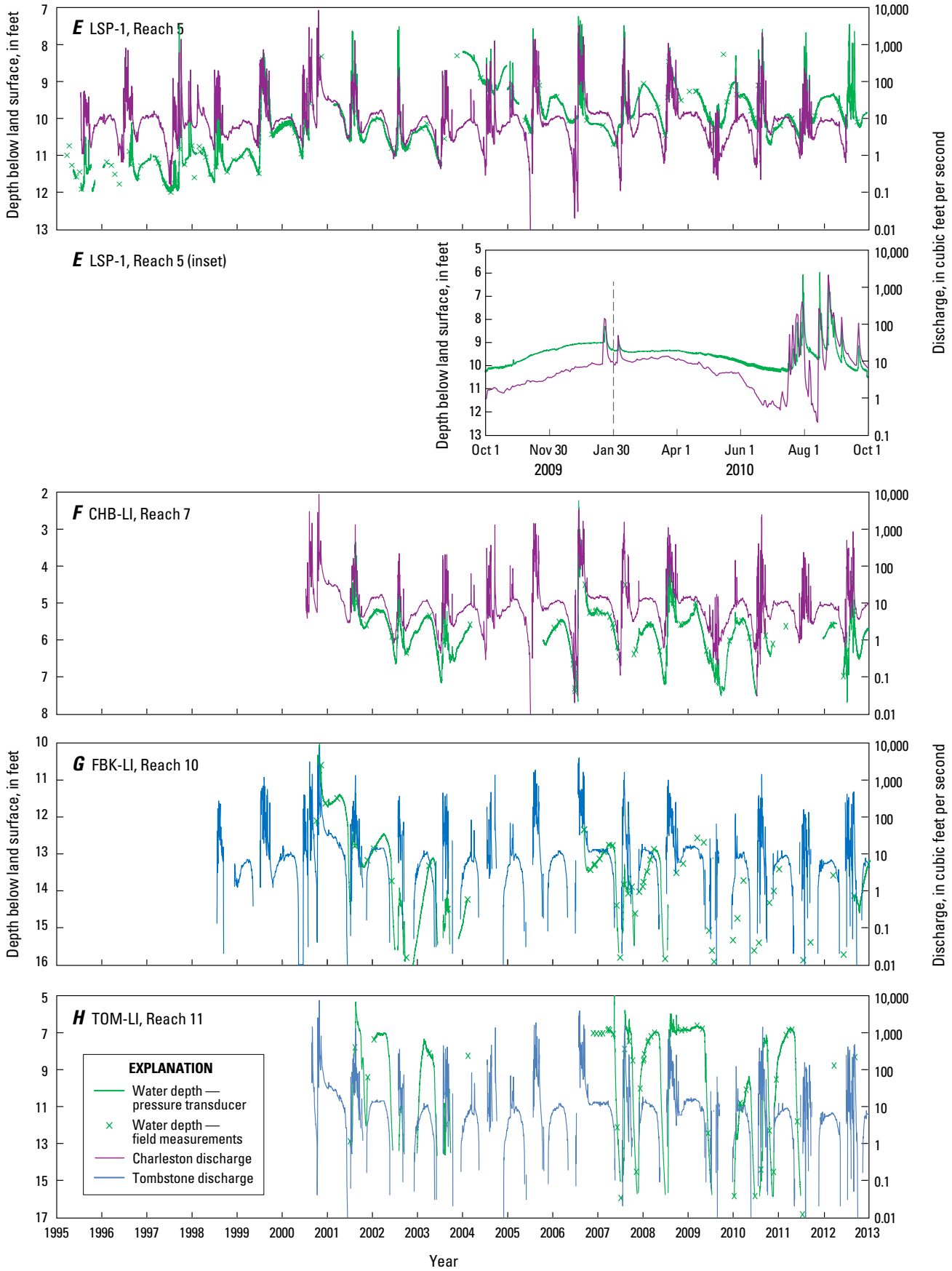
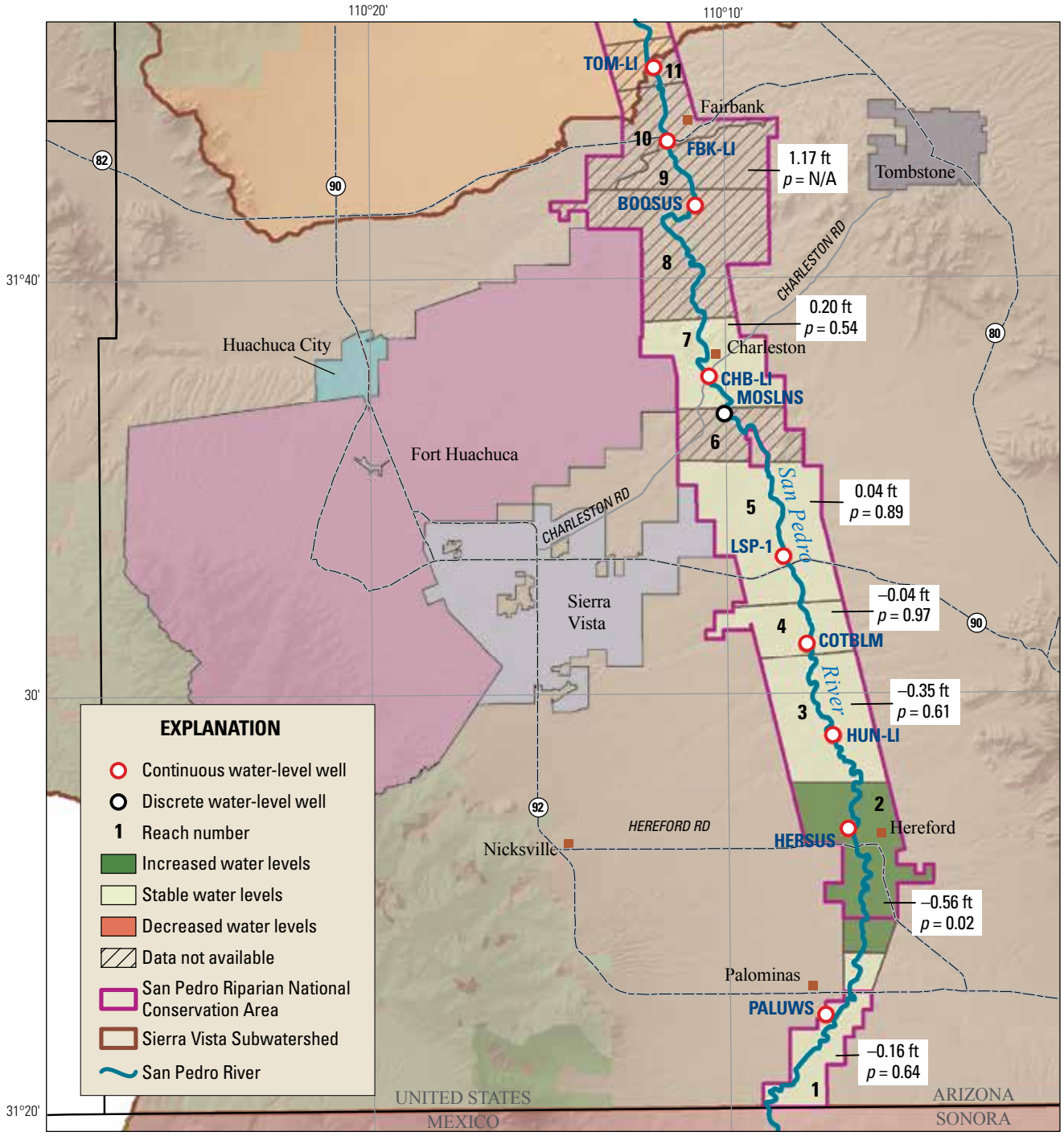


Figure 21.—Continued



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

0 1.5 3 6 MILES
0 1.5 3 6 9 12 KILOMETERS

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: -111.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Figure 22. Map showing changes in the lowest (dry season) water levels below land surface in the near-stream, alluvial aquifer along the San Pedro River in the Sierra Vista Subwatershed, southeastern Arizona. Trend is evaluated by comparing the mean of the available 2001 to 2006 data with the mean of the available 2007 to 2012 data. Black numbers in white boxes are the differences between the second half mean and the first half mean at that location, and the *p*-value from the analysis of variance (ANOVA). Negative values indicate a decrease in depth to water (a water-level rise). The Fairbank result is based on two annual data points, which was insufficient for analysis by ANOVA. ft, feet; N/A, not applicable.

1, 2, 4, and 8, wells on the post-entrenchment flood plain with sufficient data for evaluation were not available, and wells on the pre-entrenchment flood plain were used instead. Continuously recorded data were used whenever possible, augmented with manual field measurements.

In addition to visual inspection of hydrographs, lowest annual (dry season) groundwater levels were determined for the years and well locations where sufficient data existed to make this determination meaningful. Those wells were at Palominas (PALUWS), Hereford (HERSUS, reach 2), Hunter (HUN-LI, reach 3), Cottonwood (COTBLM, reach 4), Lewis Springs (LSP-1, reach 5), Charleston (CHB-LI, reach 7), and Fairbank (FBK-LI, reach 10). Analyses of data trends were accomplished by comparing the mean of the available data from the second half of the period of record (2007 to 2012) to the mean of the available data from the first half of the period of record (2002 to 2006). To provide a more robust evaluation of the dry-season water-level data, an analysis of variance (ANOVA) was then conducted that compared the earlier data to the later data. Adequate water-level data for an ANOVA were available at all wells except for FBK-LI.

Indicator 5. Near-Stream Alluvial-Aquifer Groundwater Levels—Results

Most near-stream alluvial-aquifer groundwater levels were relatively stable (fig. 21*A–H*). For example, water-level variability at PALUWS, HERSUS, and HUN-LI was similar, showing little long-term trend. Water levels in most reaches tracked streamflow, although there were differences among sites. Some alluvial-groundwater levels responded strongly to the summer monsoon peak flows, whereas others may have responded over the longer term to retirement of agricultural pumping. Lewis Springs (LSP-1, reach 5; fig. 21*E*) was one exception; water levels there have been more variable than most locations due to beaver activity (fig. 21*E*, insert; also, see indicator 7, annual fluctuation of near-stream alluvial groundwater). The highest annual water levels typically occur during the monsoon, when river stage is also high. The annual minimum tends to occur in July or during November or December.

Water-level variations in a few wells tended to be more dynamic than in most other wells. After water levels in COTBLM (fig. 21*D*) and FBK-LI (fig. 21*G*) rose following the October 2000 precipitation and flow event, they had declined more than 6 ft by 2004. Water levels then rose again, with COTBLM peaking in 2006 at about the same level as in 2000. Water levels again declined at COTBLM, but, aside from seasonal variations, remained relatively stable at FBK-LI. TOM-LI (fig. 21*H*) depth-to-water often fluctuated in excess of 10 ft annually, the most of any site, and in many years dropped below the level of the well screen and pressure transducer (around 17 ft below land surface, BLS). The contrast in the spikes in discharge and the relatively flat peaks in the groundwater data seen over some of the winter seasons may be related to seasonal saturation of the alluvial aquifer in the Tombstone gaging station area as discussed in Kennedy and Gungle (2010). In the Lewis Springs time-series plot, it is possible to resolve variability that resulted from beaver activity (fig. 21*E*, insert). This appears as a slow rise in the groundwater hydrograph relative to the streamflow hydrograph beginning in late 2009 (presumably due to local recharge from beaver ponds). After the flow event of December 24, 2010, the water level dropped abruptly (interpreted as the destruction of the beaver dam) and then tracked streamflow again.

PALUWS, HERSUS, and COTBLM were all on the pre-entrenchment flood plain, away from the riparian forest and at greater distance from the river and higher altitude than the post-entrenchment flood plain. Wells distant from the river generally have a greater depth to water than those near the river. As a result, water-level depths measured at these sites were not directly comparable to those of the Stromberg and others (2006) condition class evaluations and this evaluation overall is not directly comparable to that of Stromberg and others (2006).

Annual dry-season maximum groundwater depth throughout the SPRNCA varied less than 2 ft from 2001–02 levels, with the exception of COTBLM, where greatest annual depth to water dipped nearly 3 ft below 2001 and 2002 levels in both 2003 and 2004, and in 2012 depth to groundwater water was still more than 2 ft deeper than in 2002. Results of the annual dry-season maximum-depth analysis are shown in figure 22. Based on this analysis, sites upstream improved in the second half of the data record (2007 to 2012) relative to the first half (2000 to 2006), whereas sites further downstream have declined. Note that the Fairbank water-level results, which show the greatest decline in dry-season depth-to-water, are based on just two data years. Results of the ANOVA (valid for 6 reaches, Fairbank data insufficient) indicated that the Hereford water levels (reach 2) showed a statistically significant increase in maximum dry-season groundwater depth (p=0.02); none of the other five reaches approached statistical significance.

Indicator 6. Near-Stream Vertical Gradients—Methods

Vertical gradients are calculated from water levels in two wells screened at different depths:

$$G_v = \frac{\Delta h}{\Delta d} \tag{3}$$

where G_v is the vertical gradient, Δh is water-level elevation in the deep well minus the water-level elevation in the shallow well, and Δd is the difference in depth of the middle of the screened interval in the deep well minus the depth of the middle of the screened interval in the shallow well. As defined, a positive vertical gradient indicates a flow direction from deeper to shallower parts of the aquifer system, consistent with discharge from an aquifer system to a river. The data were grouped and analyzed using the San Pedro River reaches delineated by Stromberg and others (2006). However, only 7 of the 11 SPRNCA reaches currently have vertical-gradient well pairs, so not all are represented.

Trends referenced in this section are based on visual inspection of the near-stream vertical-gradient plots in fig. 23. A statistical approach to trend analysis was not undertaken. Such detailed analysis of these datasets would need to account for the

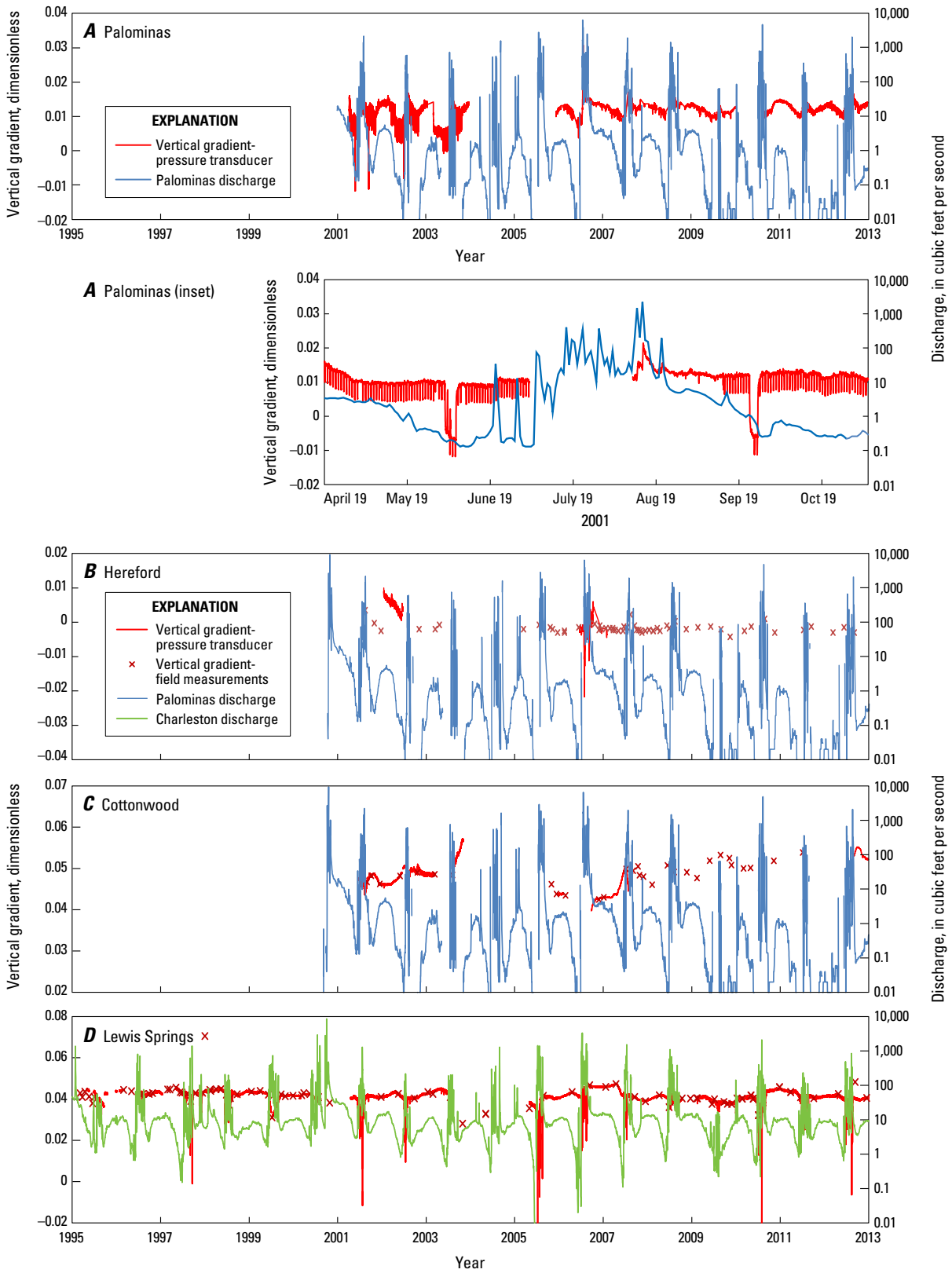


Figure 23. Graphs of vertical gradients in San Pedro Riparian National Conservation Area (SPRNCA) near-stream wells plotted with discharge at the nearest stream gage, respectively, in the Sierra Vista Subwatershed, southeastern Arizona. (*A*) Palominas, reach 1; (*B*) Hereford, reach 2; (*C*) Cottonwood, reach 4; (*D*) Lewis Springs, reach 5; (*E*) Moson, reach 6; (*F*) Boquillas, reach 8; (*G*) Fairbank, reach 10. A detailed insert of apparent nearby pumping effects on vertical gradients is included with (*A*), Palominas, reach1, and of apparent ET effects with (*E*), Moson, reach 6.

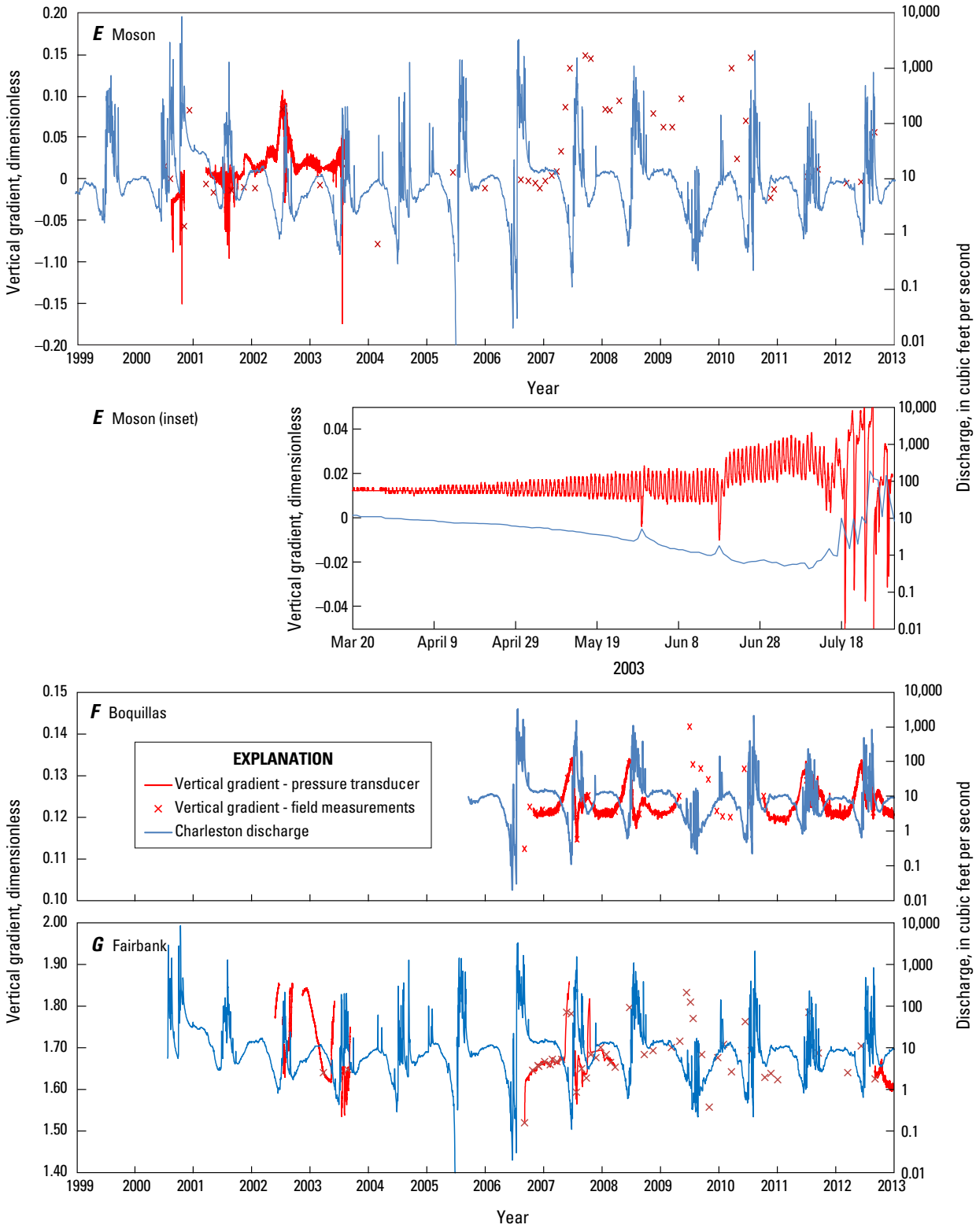


Figure 23.—Continued

complexity of measurements from different sources (manual and instrumental), widely varying time intervals between measurements, occasional large data gaps, serial correlation, and quasi-periodic seasonal variability. Such an analysis was outside the scope of the current study.

Indicator 6. Near-Stream Vertical Gradients—Results

Well pairs needed to calculate vertical gradients were available for reach 1 (PALUWS and PALUWD), reach 2 (HERSUS and HERSUD), reach 4 (COTUWD and COTBLM), reach 5 (LSP-1 and LSP-5), reach 6 (MOSLNS and MOSLND), reach 8 (BOQSUS and BOQSUD), and reach 10 (FBK-LI and FBK-LO) (table A5). With the exception of the Cottonwood site (reach 4), there were no discernible trends in vertical gradients (figs. 23*A*–*G*). With a few exceptions, as river flow increases gradients decrease. It is presumed that declines in gradient, including gradient reversals associated with large river discharges, occurred when recharge of the near-stream alluvial aquifer caused a larger magnitude and rate of water-level rise in the shallow well than in the deep well. This effect would be temporary. For the analysis of long-term trends only vertical gradients calculated during base-flow conditions were used. Responses to annual streamflow vary from site to site, such that where data are sufficient, it is clear that each site has its own particular pattern of response to streamflow variations. This pattern is usually repeated on an annual cycle, with similar trends appearing at certain wells during both drier and wetter periods. For example, compare the vertical gradients at Palominas (reach 1; fig. 23*A*) with those of Boquillas (reach 8; fig. 23*F*). Major recharge events such as occurred in October 2000 can cause short-term deflections and even reversals in otherwise stable vertical gradients. Differences in response from one site to another are likely related to a combination of factors, such as pumping, local geology (including the degree of connectivity between the regional and alluvial aquifers), and the spatial and temporal variability of beaver-pond sizes and locations.

The Palominas (reach 1) vertical gradient shows no readily discernible long-term trend (fig. 23*A*). Vertical gradients were primarily upward (positive) with a few exceptions between 2001 and 2003. In June 2001, the Palominas vertical gradient reversed abruptly (over the course of 2 hours), remained negative for about 5 days, then recovered in about 12 hours (fig. 23*A*, inset). This relatively large deflection of about 0.02 is superimposed on a diurnal fluctuation of about 0.005. Because similar fluctuations were not observed in the latter part of the record, it is likely that both were caused by nearby pumping. The last agricultural pumping in the area was retired in the mid-2000s (U.S. Department of the Interior, 2006 and 2007). Overall, vertical gradients at Palominas were slightly lower on average, and seasonal variability was greater, from 2001 to 2003 (about 0.00 to 0.015) than it was from 2005 to 2012 (about 0.10 to 0.015). These effects, too, may be a result of the earlier pumping (fig. 23*A*). From 2005 to 2012 the vertical gradient remained positive and ranged from about 0.010 to 0.015.

There were few overlapping continuous data at Hereford (reach 2) and Cottonwood (reach 4) from which to calculate vertical gradients, but vertical gradients computed from paired manual field measurements in the Hereford wells covering the period between 2001 and 2012 were slightly negative, most typically in a range from about −0.003 to 0.000. There was little overall trend in the Hereford wells' vertical gradients during the period of record. The Cottonwood wells show a comparatively large upward vertical gradient and an increasing trend since 2006 (fig. 23*C*). After peaking at about 0.57 at the end of 2003, the gradients dropped to 0.40 by the end of 2006, and subsequently steadily increased again to about 0.55 by the end of 2012.

The vertical gradient at Lewis Springs (reach 5) has the longest period of record (1995 to 2012) and showed little trend. Gradients were normally positive in the range from about 0.04 to 0.05. During and after high-flow events, vertical gradients decreased sharply, even becoming negative on five occasions, but relatively quickly (hours to days) recovered to values within the 0.04 to 0.05 range.

Moson (reach 6) vertical gradients were generally in the range of −0.05 to 0.15, with no discernable interannual patterns (fig. 23*E*) and no clear trend in the data during the period of record. For a few months in 2002 and then from 2007 to 2010, continuous and paired manual measurements, respectively, suggest the vertical gradients increased 0.05 to 1.00. The reason for these increases, particularly in the later period, is unclear. Seasonally, in some years, larger gradients occur during the summer both before and after monsoon flows. ET does appear to have a small effect on the gradient and can be seen increasing with the start of the growing season and climbing temperatures during spring and summer of 2003 (fig. 23*E*, inset).

Vertical gradients at Boquillas (reach 8) were uniformly positive and devoid of any long-term trend throughout the six years of record, ranging mostly between 0.12 and 0.14 (fig. 23*F*). Similar to Lewis Springs, during and after high-flow events at Boquillas vertical gradients decreased sharply but then returned to the more typical range for this site. However, the symmetric and repeatable nature of the inverse relation with streamflow throughout much of the year is striking and unlike any other vertical-gradient pattern in the study.

The absolute values of the vertical gradients at Fairbank (reach 10) are uncertain because the depth of the screen in the shallow well is unclear. Nonetheless, the shape and trend of the vertical gradient can still be evaluated. Fairbank gradients exhibited no clear long-term trend. Variations in the gradient do not appear to be periodic, although continuous water-level data are lacking. Gradients only occasionally correspond to variations in discharge at the Tombstone gaging station, but just as often gradients do not correspond to variations in discharge.

Indicator 7. Annual Fluctuation of Near-Stream Alluvial Groundwater—Methods

The annual fluctuation of near-stream alluvial-groundwater depths is calculated as the annual range of groundwater depths

in alluvial-aquifer wells actively monitored in each reach. This indicator is best evaluated through a continuous measurement of water levels in near-stream wells, ideally with pressure transducers recording water-level depth at least every 6 hours. For Stromberg and others' (2006) study, fluctuations were calculated as the difference between minimum winter and maximum summer flood-plain-weighted depths across well transects (fluctuations were smaller near the river and greater near the flood-plain perimeter). In the current report, such calculations were not possible due to the reduction in the number of SPRNCA wells being monitored continuously in each transect. When possible, wells situated on the post-entrenchment flood plain, typically within the cottonwood-willow forest, were used to evaluate conditions representative of the riparian forest in that reach. For four of the reaches, wells on the post-entrenchment flood plain with sufficient data for evaluation were not available, so wells on the pre-entrenchment flood plain were used instead (PALUWS, HERSUS, COTBLM, and BOQSUS).

Continuous groundwater-level measurements were used whenever available, and supplemented by the discrete, manual field measurements. Winter peak-groundwater-level data were selected from the month when water levels were highest and most stable. Typically the highest winter water levels occurred in March, but over the period of record, highest winter water levels occurred in all months from November to April. The annual dry-season low-water level was selected from the months of May through September. Typically the lowest levels occurred in June or July, but in years of less summer rain, such as 2009, the low did not occur until September at some locations.

The wells available for evaluation of annual fluctuation of near-stream alluvial groundwater were PALUWS (reach 1), HUN-LI (reach 3), COTBLM (reach 4), LSP-1 (reach 5), and CHB-LI (reach 7) (table A5). No wells downstream of Charleston had datasets adequate for evaluation of annual fluctuation. Because wells distant from the river generally have smaller fluctuations in water levels than those near to the river, they are not directly comparable. Analysis of trends in annual fluctuations of individual wells completed in the alluvial aquifer was accomplished by comparing the means of the available data from the second half of the period of record (2007–12) to the first half of the period of record (2002–06). To provide a more robust evaluation of the annual water-level-fluctuation data, an ANOVA was conducted that compared the earlier data to the later data. Adequate water-level data for an ANOVA were available at five of the six wells.

Indicator 7. Annual Fluctuation of Near-Stream Alluvial Groundwater—Results

From 2001 to 2012 there are 7 years of intermittent data available for PALUWS, reach 1 (Palominas); all annual fluctuations were less than 1.1 ft (fig. 24) with a median value of about 0.8 ft. Sufficient data for HERSUS, reach 2 (Hereford), were only available for 2007 and later; fluctuations were less than 1 ft for the period. Annual fluctuations at HUN-LI, reach 3 (Hunter), were about 1.9 to 2.6 ft. Annual fluctuations in COTBLM, reach 4 (Cottonwood), were 1.5 to 2.0 ft in most years, although water levels there fluctuated just 1.09 ft in 2001, in 2010 1.23 ft, and nearly 2.5 ft in 2007. At LSP-1, reach 5 (Lewis Springs), fluctuations were fairly stable from 2001 to 2005 but became more variable in subsequent years. Since 2006, water levels have occasionally undergone a slow rise with abrupt declines that appear related more to beaver dams in the Lewis Springs area than to trends in streamflow. The slow rise occurs as water backs up behind the dam, and the rapid decline follows when a high-flow event destroys the dam (fig. 23*E*). Fluctuations in CHB-LI, reach 7 (Charleston), were consistently greater than about 1.3 ft beginning in 2006, although in 2012 water levels fluctuated less than 1.0 ft. BOQSUS, reach 8 (Boquillas), water-level fluctuations increased from 1.38 in 2007 to 1.66 ft in 2012.

Results of the annual water-level fluctuation analysis are shown in figure 25. Stromberg and others (2006) observed that increases in alluvial-groundwater fluctuations, like decreases in alluvial-groundwater levels, lead to a decrease

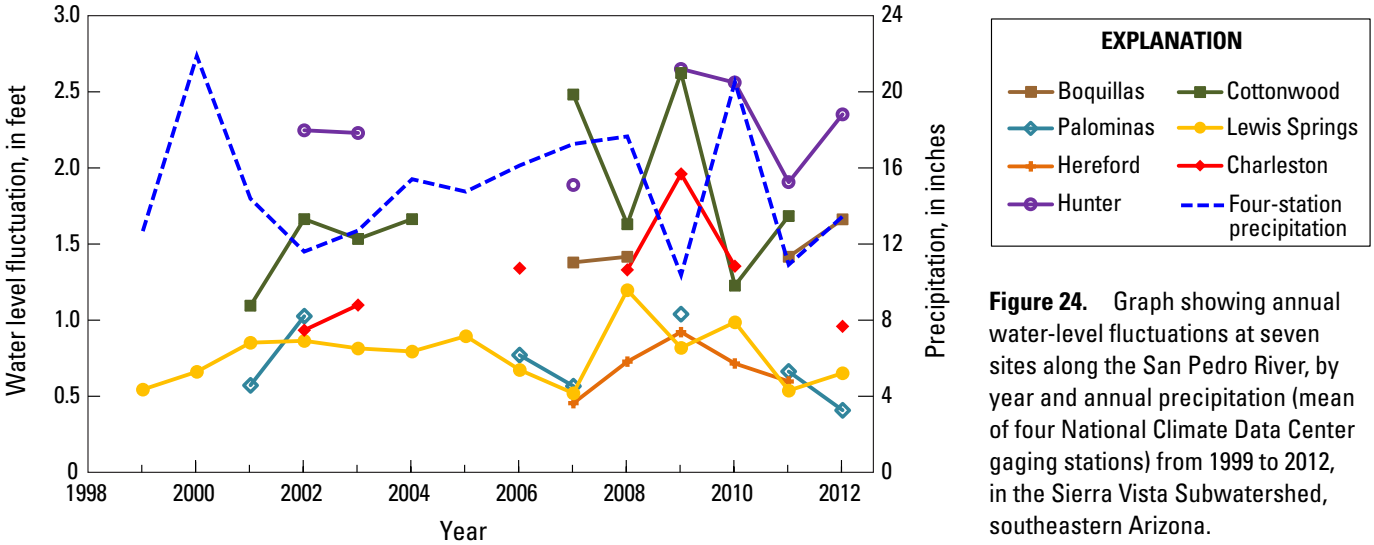
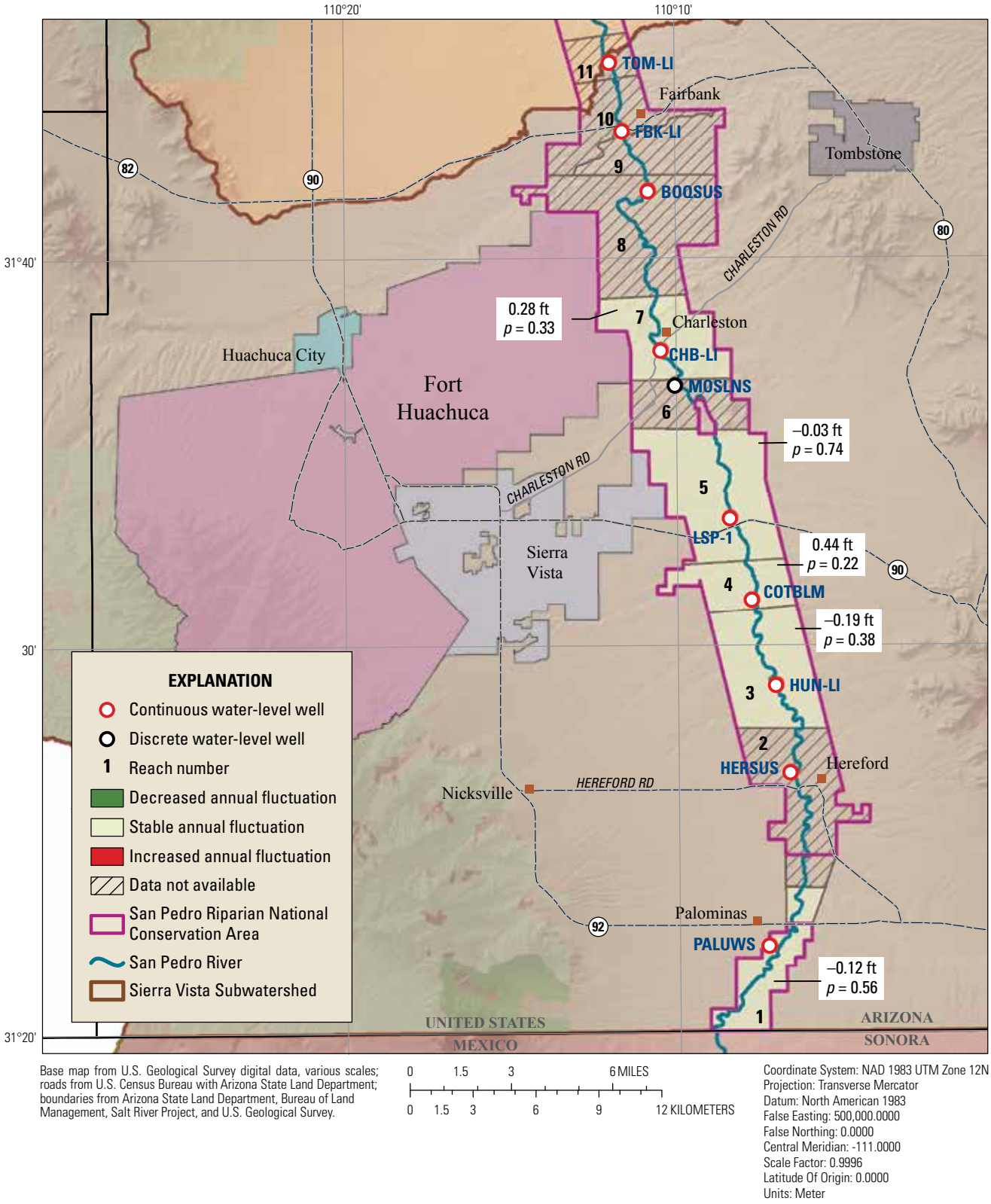


Figure 24. Graph showing annual water-level fluctuations at seven sites along the San Pedro River, by year and annual precipitation (mean of four National Climate Data Center gaging stations) from 1999 to 2012, in the Sierra Vista Subwatershed, southeastern Arizona.



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: -111.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Figure 25. Map showing changes in the annual water-level fluctuation in the near-stream, alluvial aquifer along the San Pedro River, in the Sierra Vista Subwatershed, southeastern Arizona. Trend is evaluated by comparing the mean of the available 2001 to 2006 data with the mean of the available 2007 to 2012 data. Black numbers in white boxes are the differences between the second half mean and the first half mean at that location, and the *p*-value from the analysis of variance (ANOVA). Negative values indicate a decrease in annual water-level fluctuation. ft, foot.

in cottonwood and willow abundance. Alluvial water-level fluctuations show a pattern similar to the dry-season alluvial water levels (indicator 5), with increased groundwater fluctuations at sites further downstream. However, results of the ANOVA indicate that the changes in fluctuation in the five wells were not statistically significant.

Group 3. San Pedro River Indicators

Indicator 8. Streamflow Permanence—Methods

Stromberg and others (2006) found that streamflow permanence explains more of the variance in the basal area of cottonwoods and willows than either groundwater depth or groundwater fluctuation. Interannual variability of this indicator was analyzed as a function of Julian day for 10 sites, from 2002 to 2012. Of the 11 reaches in the Sierra Vista Subwatershed, two have been without streamflow permanence monitoring since 2003 (reach 4 and reach 9) and one, reach 8, includes both the Boquillas and Charleston Mesquite permanence monitoring sites (fig. 26). From July 2000 through September 2003, Leenhouts and others (2006) assessed flow permanence from data collected at numerous sites along the San Pedro River using temperature sensors buried 0.3 m below the streambed, electrical-resistivity sensors at the streambed surface, manual periodic-discharge measurements, and stream-stage recorders.

BLM, and more recently ARS, assessed streamflow permanence from images collected with automatic remote digital cameras mounted in trees at six locations within the subwatershed. Flow condition was assessed as either “flow” or “no-flow” (unmoving, ponded water is coded “flow”). The photographic data were combined with stage data from the three mainstem USGS stream gages (Palominas, Charleston, and Tombstone) and the one stage recorder (Lewis Springs) for a total of 10 streamflow-permanence data collection sites in the SPRNCA (fig. 26). The earlier, Leenhouts and others (2006) data were collected at different locations than the stream stage and photography sites and so were not directly comparable and were not included here. Only the stream-stage data and photographic data were evaluated. Flow-permanence data were evaluated daily (flow or no-flow) and are shown in figure 27 by year, 2007–12. Data are also plotted as percent of year with flow versus year (fig. 28).

Indicator 8. Streamflow Permanence—Results

Figure 27 provides an overview of the annual timing and extent of no-flow periods from 2007 to 2012. Data gaps impede analysis in some cases, especially in 2011 (fig. 27). Hereford (reach 2), Lewis Springs (reach 5), Moson (reach 6), Charleston (reach 7), and Boquillas (reach 8) maintained perennial flow throughout the period of record with a few brief exceptions. Palominas (reach 1), Hunter (reach 3), Charleston-Mesquite (reach 8), Fairbank North (reach 10), and Tombstone (reach 11) all had significant periods of no flow, typically during June and July with occasional dry periods during the late fall. The driest years of the period were 2009 and 2011 (fig. 28), and this is reflected in the long periods of mostly no flow at Fairbank and Tombstone (fig. 27).

Figure 28 plots total days of flow for each site by year (years with greater than 10 percent missing data have been removed), along with annual precipitation. Visual inspection shows that flow permanence tracks precipitation at nearly every site with intermittent flow for the 5 to 10 years of flow-permanence data currently available.

Indicator 9. Base Flow on San Pedro and Babocomari Rivers—Methods

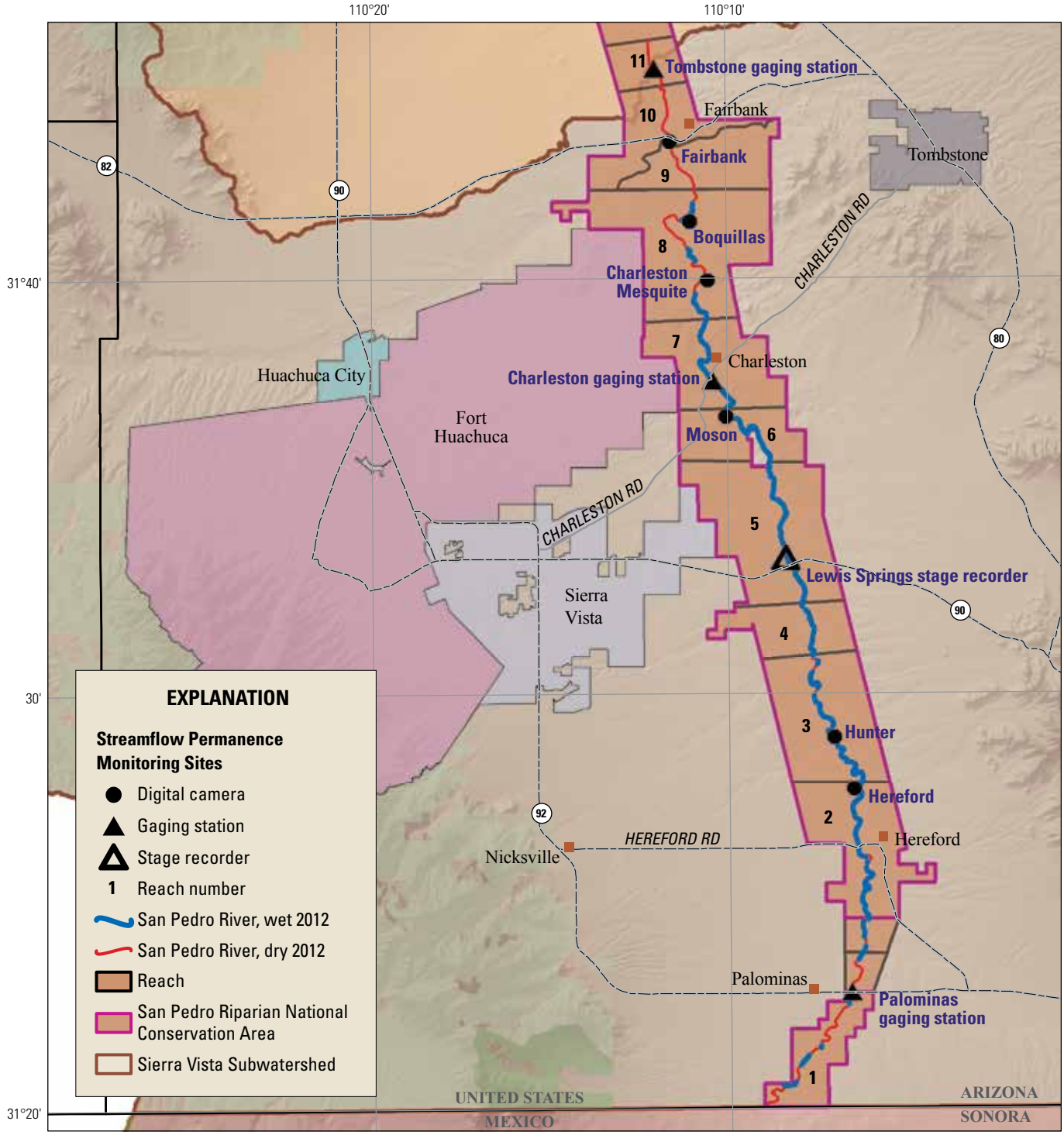
Base flow was estimated for four existing gaging stations—Palominas, Charleston, Tombstone, and the Lower Babocomari (fig. 20)—and using 2 different methods (3-day low flow and winter base flow). Because the period of record for most of the streamflow gaging stations is much longer than for many of the other indicators, base-flow trends have been evaluated over both the entire period of record and a shorter time period (12 years).

Three-Day Low Flow

The 3-day low flow is the lowest flow averaged over three consecutive days for a given period, and is commonly used under the assumption that the lowest streamflow during the year is representative of base flow (Wahl and Wahl, 1988). For the San Pedro River, the annual 3-day low flow typically occurs in late June and is heavily influenced by the amount of ET. At all stations evaluated except Charleston, annual 1-, 3-, and 7-day low flows were usually zero. January 3-day low flow, which is largely independent of ET, is used instead to represent base-flow conditions. January 3-day low flow is shown in figure 29 for Palominas, Charleston, Lower Babocomari, and Tombstone. Annual 3-day low flow for Charleston is shown in figure 30.

Winter Base Flow

Base flow can be difficult to estimate for streams where the base-flow period begins or ends with periods of zero-flow. The delta method is a hydrograph-separation technique developed specifically for the San Pedro River (Kennedy and Gungle, 2010). Base flow is determined using a daily discharge-difference threshold. The delta filter uses an increase or decrease in mean daily flow from the previous day to indicate base flow on that day. For the delta method, if the increase or decrease in mean daily flow is greater than the day before by a specified, optimal threshold (2 cubic feet per second, ft^3/s, in this case), that day is considered storm flow-influenced, and the base-flow



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

0 1.5 3 6 MILES
0 1.5 3 6 9 12 KILOMETERS

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: -111.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Figure 26. Map showing location of streamflow-permanence monitoring sites along the San Pedro River in the Sierra Vista Subwatershed, southeastern Arizona. June 2012 wet-dry data are included for reference (see "Indicator 10. Wet-Dry Status" in text).

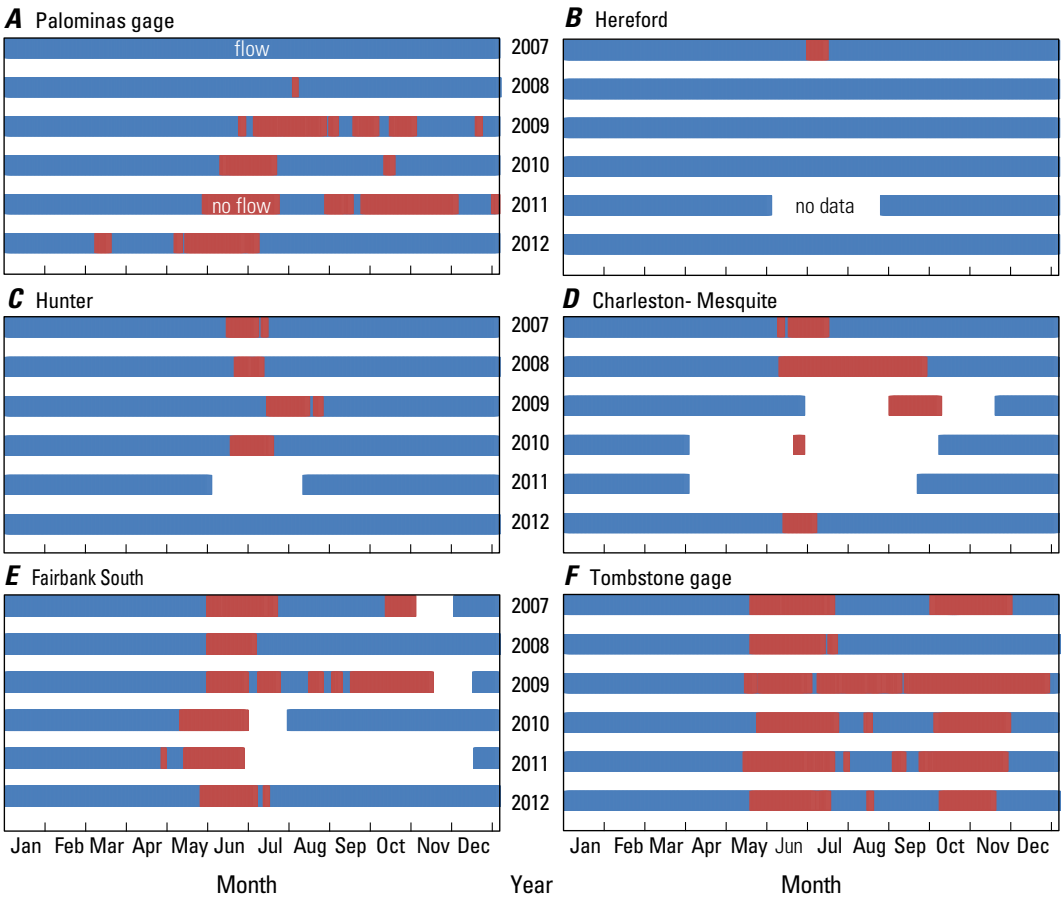


Figure 27. Graphs showing streamflow permanence as a function of month and year at sites along the San Pedro River in the Sierra Vista Subwatershed, southeastern Arizona. *A*, Palominas gage; *B*, Hereford; *C*, Hunter; *D*, Charleston-Mesquite; *E*, Fairbank South; *F*, Tombstone gage. Lewis Springs, Charleston, Moson, and Boquillas sites were perennial for period of record and are not shown. Blue indicates flow, red indicates no flow, and white indicates no data available.

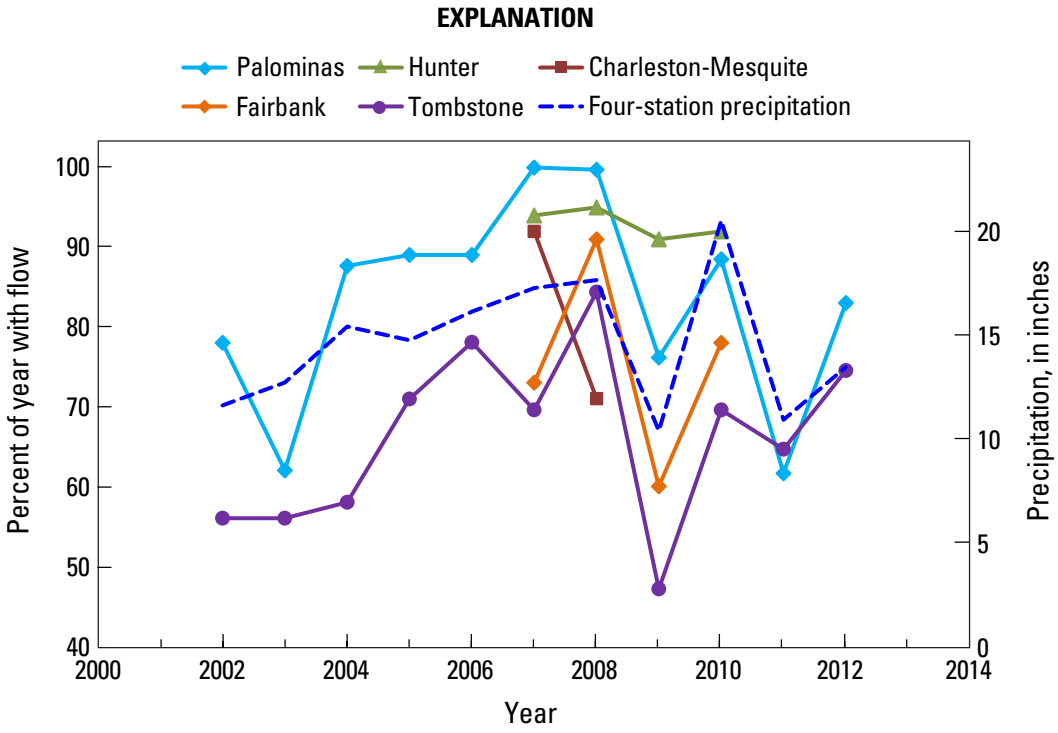


Figure 28. Graph showing San Pedro River annual streamflow permanence and annual precipitation (mean of four National Climate Data Center precipitation gaging stations) for 2002–12, in the Sierra Vista Subwatershed, southeastern Arizona. Stations with 100-percent flow permanence are not shown (Hereford, Hunter, Lewis Springs, Moson, Charleston, Boquillas).

rate is linearly interpolated between the two nearest days of base flow discharge (Kennedy and Gungle, 2010).

After determining base flow from hydrograph separation, the total volume of winter base flow can be calculated by integrating the hydrograph over a given period. For the Tombstone station, the winter base-flow season most commonly begins and ends with periods of zero flow, which provides a specific time period over which to integrate streamflow (Kennedy and Gungle, 2010). At Palominas, Charleston, and Lower Babocomari gaging stations, streamflow was often continuous in the fall and spring, making it more difficult to determine the start and end dates of winter base-flow discharge. Therefore, the volume of base flow at each stream-gaging station was calculated using the fixed period from December 1 to April 30 of each water year. This represents the period when streamflow was least-influenced by ET and most representative of base-flow conditions. Although the resultant base-flow volume was smaller than the entire volume of winter base flow, it is nonetheless useful and has been used here to evaluate trends over time.

Indicator 9. Base-Flow Discharge on San Pedro and Babocomari Rivers—Results

Both of the low flow indicators—3-day low flow and base flow-separated flow—show a similar pattern of decline at all four gaging stations during the respective periods of record. The difference between January 3-day annual low flow and annual 3-day low flow is primarily controlled by seasonal precipitation; ET demand in late spring tends to have more of an effect on annual 3-day low flow than on January 3-day low flow (figs. 29, 30). At Lower Babocomari and Tombstone gaging stations, a decline in January 3-day low flow is apparent. At the Palominas gaging station the decline in January 3-day low flow is less apparent, in part because there were years when January 3-day low flows approached zero throughout the period of record. However, there were more years with relatively high January 3-day low flow (greater than 5 ft^3/s) during the earlier part of record than during the latter part (fig. 29).

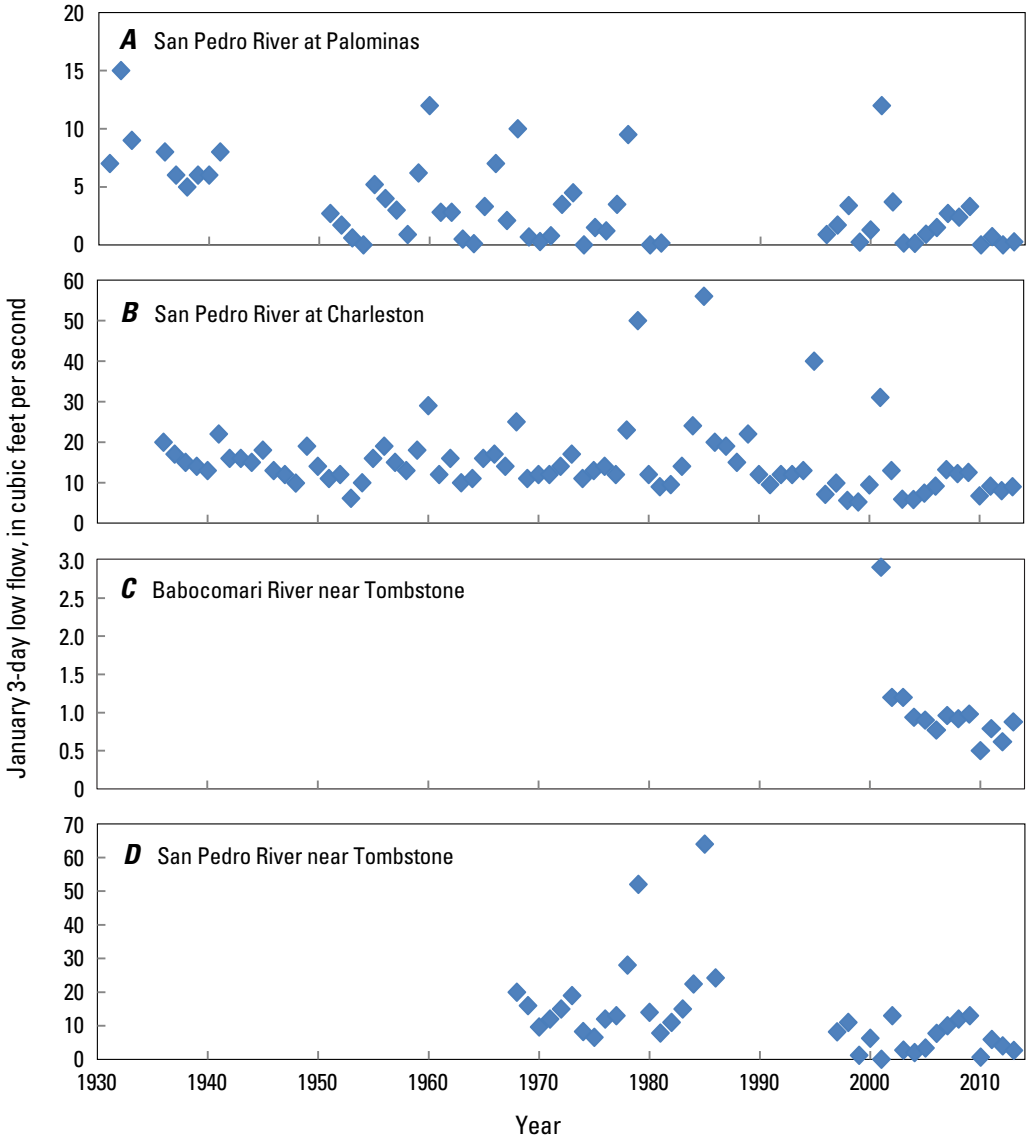


Figure 29. Graphs showing January 3-day low flows at the (*A*) San Pedro River at Palominas (09470500), (*B*) San Pedro River at Charleston (09471000), (*C*) Babocomari River near Tombstone (094714000), and (*D*) San Pedro River near Tombstone (09471550) gaging stations in the Sierra Vista Subwatershed, southeastern Arizona.

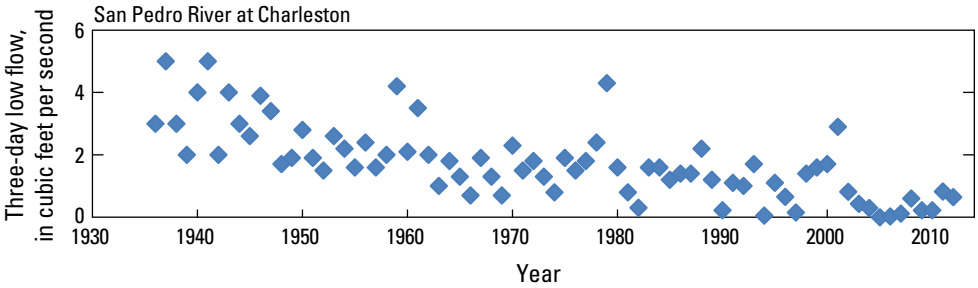


Figure 30. Graph showing annual (mainly summer) 3-day low flows at San Pedro River at Charleston (09471000) gaging station in the Sierra Vista Subwatershed, southeastern Arizona.

Base flow-separated flow is fairly uniform throughout the period of record, with exceptionally large base flows only occurring in years with large floods, such as 1978 and 1984 (fig. 31). From 1905 to 1995, median base flow-separated flow at Charleston was 15.3 ft^3/s; from 1996 until 2013 median base flow-separated flow is 0.6 ft^3/s. Base flow-separated flow is relatively stable between the 1940s and early 1970s and again from the mid 1990's to present but at a lower magnitude. The transition between these more stable periods is marked by several years with large base flows in the late 1970s and 1980s. Base flow-separated flow at Tombstone gaging station is nearly identical to that at Charleston during years with relatively high base flow-separated flow; in other years Tombstone has slightly less base flow-separated flow than Charleston (fig. 31). The relatively short period of record at the Lower Babocomari gaging station exhibits a decrease in base flow-separated flow during the 2000s; median base flow-separated flow for 2002–04 is 1.1 ft^3/s, and for 2011–13 period it is 0.82 ft^3/s. Base flow-separated flow at Palominas gaging station is much more variable than at the other stations, yet still shows a decline from the early part of the record (1951–81; median base flow-separated flow=2.6 ft^3/s) to the latter part (1996–2013; median base flow-separated flow=1.3 ft^3/s).

Base flow has been declining at the Palominas, Charleston, Tombstone, and Lower Babocomari gaging stations over the entire period of record. Although near-stream vertical-gradient data provided no indication that the reduction in base flow was the direct result of groundwater pumping (Kennedy and Gungle, 2010; see also "Indicator 5. Near-Stream Vertical Gradients"), a positive trend in the stable isotope data at the Palominas and Lower Babocomari gaging stations (see "Indicator 12. Stable Isotopes") suggested that a decreasing proportion of groundwater contributing to streamflow at these locations could be contributing to the reduction in base flow. In addition, groundwater flow

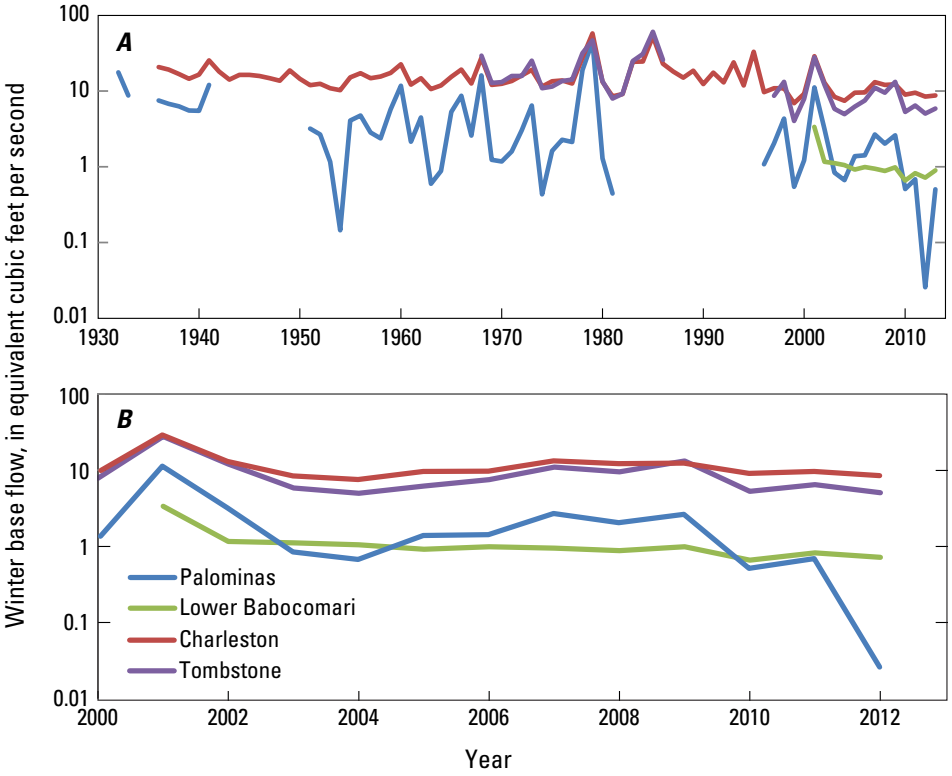


Figure 31. Graphs showing base flow-separated flow for San Pedro River at Palominas (09470500), San Pedro River at Charleston (09471000), Babocomari River near Tombstone (094714000), and San Pedro River near Tombstone (09471550) gaging stations in the Sierra Vista Subwatershed, southeastern Arizona. On y-axis, "equivalent cubic feet per second" is the volume of base flow-separated flow divided by the period of calculation, January 1 to April 30, inclusive. *A*, Period of record at each stream-gaging station; *B*, 2000 to 2012.

modeling, which can isolate the effects of groundwater pumping, has shown that water levels in the subwatershed have declined since 1902, reducing the groundwater gradients that influence groundwater flow toward the river by as much as 17 percent (Lacher and others, 2014). Water-level declines also reduce the total volume of water that flows to the river.

Indicator 10. Wet-Dry Status—Methods

Mapping of wet-dry status, undertaken during the driest part of the year (third weekend in June), uses a 1-day, on-the-ground visual evaluation of the entire Sierra Vista Subwatershed stream reach to determine wetted stream length. The wet-dry mapping fieldwork was done by volunteer citizen scientists who surveyed assigned river sections on foot or horseback. The volunteers recorded the start and end points of each wetted length using hand-held GPS units. The resulting coordinates were mapped as linear segments of wetted stream in a geographic information system (GIS) program (more details are provided in Turner and Richter, 2011, and at http://azconservation.org/downloads/san_pedro_wet_dry_mapping). Lengths of all of the wetted reaches were summed to determine total wetted length. Turner and Richter (2011) evaluated the 1999 to 2010 wet-dry data in 5 mi lengths, but for the purposes of this report data for 1999–2012 were evaluated based on the variable-length river reaches identified by Stromberg and others (2006). Statistical significance of linear temporal trends was evaluated using the nonparametric Mann-Kendall test after removing the 1-year autocorrelation. The trend test was evaluated using a significance level of 0.05 (the null hypothesis that the correlation has occurred at random is rejected if $p \leq 0.05$, in which case it is instead concluded that a statistically significant trend exists).

Indicator 10. Wet-Dry Status—Results

Wet-dry status varies by location (figs. 32 and 33). Although Palominas (reach 1), Charleston Mesquite (reach 8), and Fairbank North (reach 10) trended down from 1999 to 2012 and Hereford (reach 2) and Hunter (reach 3) trended up (fig. 33), results of the Mann-Kendal test show a significant linear trend only for Hereford (p=0.005), where there has been a fairly steady increase in wetted length. This may be a response to the retirement of agricultural pumping upstream in the Palominas area (U.S. Department of Interior, 2007, 2008). Lewis Springs (reach 5), Moson (reach 6), and Charleston (reach 7) were consistently wet through all or most of their lengths, whereas Fairbank South (reach 9) and Tombstone (reach 11) were consistently dry throughout most of their lengths since at least 2004. Cottonwood (reach 4), showed little overall trend toward a drier or wetter late June.

Indicator 11. San Pedro River Water Quality—Methods

The San Pedro River water-quality indicator comprises a suite of water-quality analytes and parameters that have been monitored long-term in on the San Pedro River. As a means to summarize and quantitatively evaluate water-quality condition and trends, analytes measured at the stream-gaging station San Pedro River at Charleston were evaluated in the context of the Arizona Department of Environmental Quality's (ADEQ) water-quality standards. The two standards evaluated were the drinking-water standard and the chronic "warm" water aquatic and wildlife standard (plants and animals using water occurring at altitudes less than 5,000 ft for habitation, growth, or propagation). Nutrient standards and criteria vary throughout the State and pertain mostly to lakes and reservoirs and a few select rivers. Warm-water stream criteria for aquatic organisms have not been fully established in Arizona, so the Environmental Monitoring and Assessment Program (EMAP) ecological thresholds established by Arizona Game and Fish Department, USGS, and U.S. Environmental Protection Agency (EPA) were used here for some analyte and nutrient criteria to provide perspective (Stoddard and others, 2005; Robinson and others, 2006).

Water-quality data have been collected from the San Pedro River at the Charleston stream-gaging location since 1964, and the location has been a long-term NAWQA reference site since 1991, although the frequency and timing of collection has been variable. Several locations along the river were sampled for various water-quality constituents over the years, but none have a long-term record comparable to the Charleston location. Several hundred samples have been collected at the San Pedro River at Charleston stream-gage site since 1964 in accordance with USGS field collection guidelines and more recently with protocols outlined in the USGS National Field Manual (U.S. Geological Survey, variously dated). These samples were collected for a variety of purposes and at a variety of sampling intervals and analyzed for a variety of analytes. Since the 1990s all samples were collected with nationally consistent protocols as part of NAWQA (Gilliom and others, 1995).

To better understand the trends in the water-quality analytes of interest, the analytes were analyzed on the basis of four components—the relation to discharge, seasonality, long-term trend, and a random component using the Exploration and Graphics for River Trends (EGRET-WRTDS) tool (Hirsch and De Cicco, 2014). EGRET is a USGS statistical program used to investigate significant constituent trends. This modeling process allows the user to estimate the concentration for any day or discharge used in the period of analysis. It produces tabular and graphical representations of the concentration and the flux (or load) of the analyte over time using multiple linear regression techniques (Hirsch and De Cicco, 2014). The analysis works best on larger datasets representing a wide range of conditions and discharge. Unlike previous tools, the EGRET-WRTDS tool makes it possible for the analysis to incorporate censored values. Because of these constraints, only analytes that showed significant monotonic trends in concentration, as indicated by the Mann-Kendall analysis, and had sample sizes greater than 100 (of which more than half were detections) were analyzed using the EGRET-WRTDS tool. Those analytes were sulfate, orthophosphate, total phosphorus, and total suspended sediment concentrations.

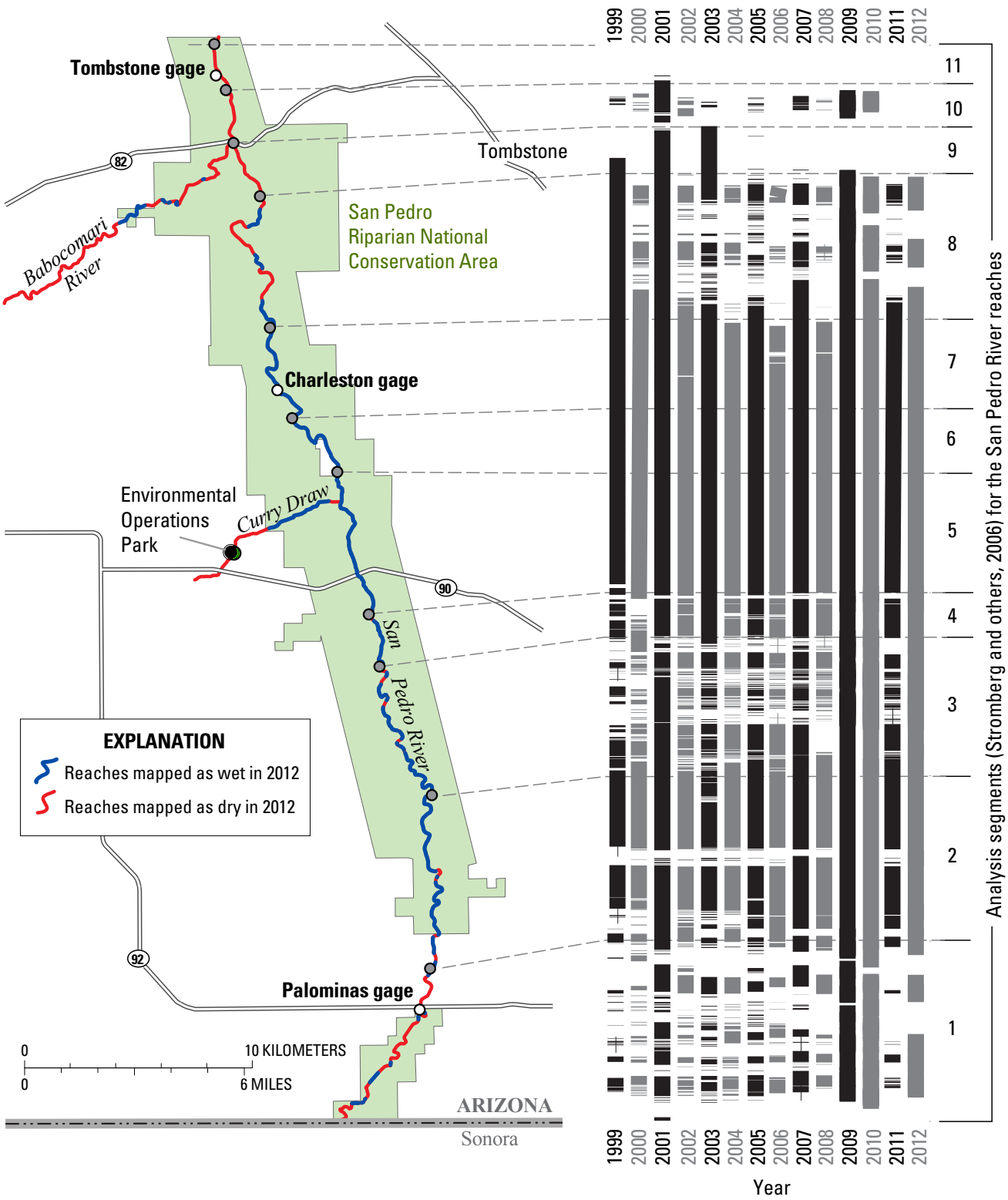


Figure 32. Map with illustration showing wet-dry status results from the San Pedro Riparian National Conservation Area in the Sierra Vista Subwatershed, southeastern Arizona. The dark-blue river line shows reaches that were mapped as wet in 2012. The red river line shows reaches that were mapped as dry in 2012. The vertical, occasionally broken gray bars on the right side represent wet reaches found in the third week of June each year, 1999–2012; white spaces represent dry reaches found in the third week of June each year, 1999–2012. Labels on the far right identify the 11 analysis segments, which correspond to Stromberg and others' (2006) San Pedro River reaches.

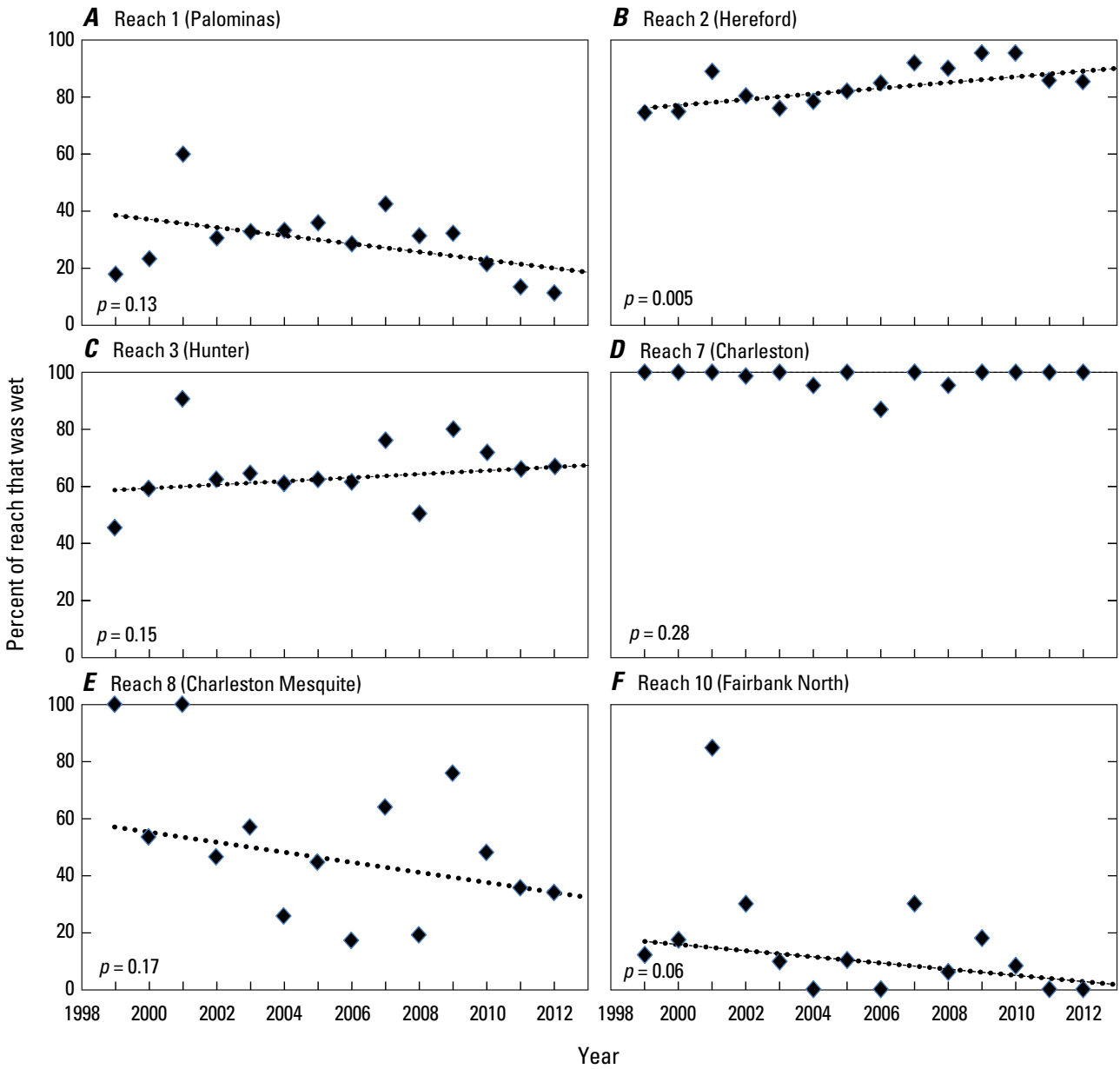


Figure 33. Graphs showing annual wetted lengths for 6 of the 11 reaches of the San Pedro River in the Sierra Vista Subwatershed, southeastern Arizona, shown in figure 32 as percent of total reach length. *A,* reach 1 (Palominas); *B*, reach 2 (Hereford); *C*, reach 3 (Hunter); *D*, reach 7 (Charleston); *E*, reach 8 (Charleston Mesquite); *F*, reach 10 (Fairbank North). Estimates of linear trend (regressions) are shown as dashed lines (nonparametric Mann-Kendall test). Hereford (reach 2) is the only reach that showed a statistically significant trend.

Much of the trace-element and nutrient data collected at the San Pedro River was reported as "censored" or "not detected." This was because the concentration of the analyte was often below the analytical capabilities of the laboratory and so could not be quantified with any statistical confidence. When conducting statistical analyses, censored data must be treated differently than exact values and are reported as less than a threshold that represents a minimum detectable concentration that the laboratory deems statistically valid for that constituent.

Indicator 11. San Pedro River Water Quality—Results

Water-Quality Parameters and General Chemistry (Major Ions)

Water-chemistry parameters (temperature, specific conductance, pH, dissolved oxygen) at the San Pedro River at Charleston were relatively stable between water years 1987 and 2012, and the parameters did not show any significant trends over the period of record (fig. 34 and table 4). The physical basin characteristics and geographic location of the contributing watershed to the Charleston gage location create conditions that cause water-chemistry parameters to naturally vary widely throughout the year. The low elevation and low gradient of the watershed combined with a mostly wide and shallow channel morphology contribute to the high seasonal and diel (24 hour period) variability of parameters. The conditions can be extreme, ranging from periods of sustained flow during wet winter months to parts of the year with no in-channel flow, followed by rapid onset of bank-full flow during summer convective storms. This large variability can be seen in the wide range of parameter observations, where the maximum is 5 to 10 times greater than the minimum for most water years. Another factor that can introduce variability into the dataset is the time of day the samples were measured or collected.

The mean water temperature at the Charleston gage is 18 degrees Celsius (°C), but water temperatures ranged from less than 5 °C in January to more than 30 °C in July. Temperature is a significant factor that influences other water-quality parameters, such as dissolved oxygen concentrations, which are generally inversely related to temperature. Dissolved oxygen fluctuates throughout the day depending on biologically mediated consumption primarily through respiration and photosynthesis. The mean concentration of dissolved oxygen was 8.43 milligrams per liter (mg/L). Although concentrations infrequently dropped below the ADEQ warm-water aquatic life criterion of 6 mg/L, no apparent patterns or trends were evident. The mean pH of the Charleston location was 8.32 standard units, which is moderately basic (pH>7 standard units). No exceedances of the ADEQ warm-water aquatic life criterion were observed at the Charleston location. Specific conductance is an indirect measure of the dissolved solids in water. The mean specific conductance was 484 microsiemens per centimeter (µS/cm) at 25 °C. The ADEQ warm water criterion is 1,000 µS/cm, and only one measurement exceeded the criterion during the period of analysis. Specific conductance was the parameter most obviously influenced by increased discharge. Plots of discharge (x) versus specific conductance (y) exhibit a slope reversal (from positive to negative) at about 50 to 60 ft^3/s. In general, conditions observed at the Charleston gage were acceptable or favorable to aquatic biota present in this stream reach.

The major-ion concentrations indicate the stream at the Charleston location was a calcium/mixed cation bicarbonate water (fig. 35). Most samples had percentages between 10 and 30 percent of chloride and fluoride. The sulfate composition was slightly higher in some samples but was still less than 40 percent. Samples collected in the months of November through March showed a higher composition of magnesium, sodium, and bicarbonate (fig. 35). The samples collected in months associated with convective summer storms and in the fall had a greater percentage of calcium and lower proportions of the ions listed for the winter and spring months. Season and the related flow-regime influence the major-ion water chemistry. During high-flow months, many major ions such as fluoride and bicarbonate had decreases in concentrations; potassium and calcium instead increased during these high-flow months. Significant upward trends over the period of record were observed in a few individual cations, such as magnesium (τ,128; $p<0.0371$), sodium (τ, 0.125; $p<0.0382$) and bicarbonate (τ, 0.164; $p<0.002$), but the strength of the trends was weak (table 4, available online only as an Excel (.xlsx) table at https://doi.org/10.3133/sir20165114).

Fluoride (table 4 and fig. 36) showed a significant upward trend (τ=0.216, $p<0.0003$), but most of the concentrations from 1993 to 2012 were between 0.2 and 0.8 mg/L, indicating that changes were likely to be minimal with regard to impacts to the stream reach. Downward trends were observed in chloride and sulfate concentrations. Similar to fluoride, chloride concentrations were low, and the strength of the trend was weak (τ,−0.106; $p<0.0428$). The downward trend in sulfate concentrations (τ,−0.263; $p<0.0001$) was much more prominent, decreasing by more than half between water years 1987 and 2012 (fig. 36).

Sulfate was one of the analytes that changed the greatest over the period of analysis (fig. 36). Sulfate concentrations decreased about 32 mg/L between 1978 and 2012, and between 1986 and 2012 sulfate concentrations decreased on average about 0.94 mg/L per year, which corresponds to a mass flux of about 72 metric tons per year (t/yr) of sulfate. The greatest decrease (slope percent per year) occurred between 2003 and 2012 (about a 3-4 percent per year change in concentration). There were some elevated mean concentration and flux observations between 1978 and 1985 that could have been related to periodic mine releases, in the mid-1980s, from the Cananea Copper Mine (King and others, 1992) that resulted in sulfate contamination and localized fish kills from the acidic pH. Overall concentrations were low compared to the drinking-water secondary maximum contaminant level of 250 mg/L.

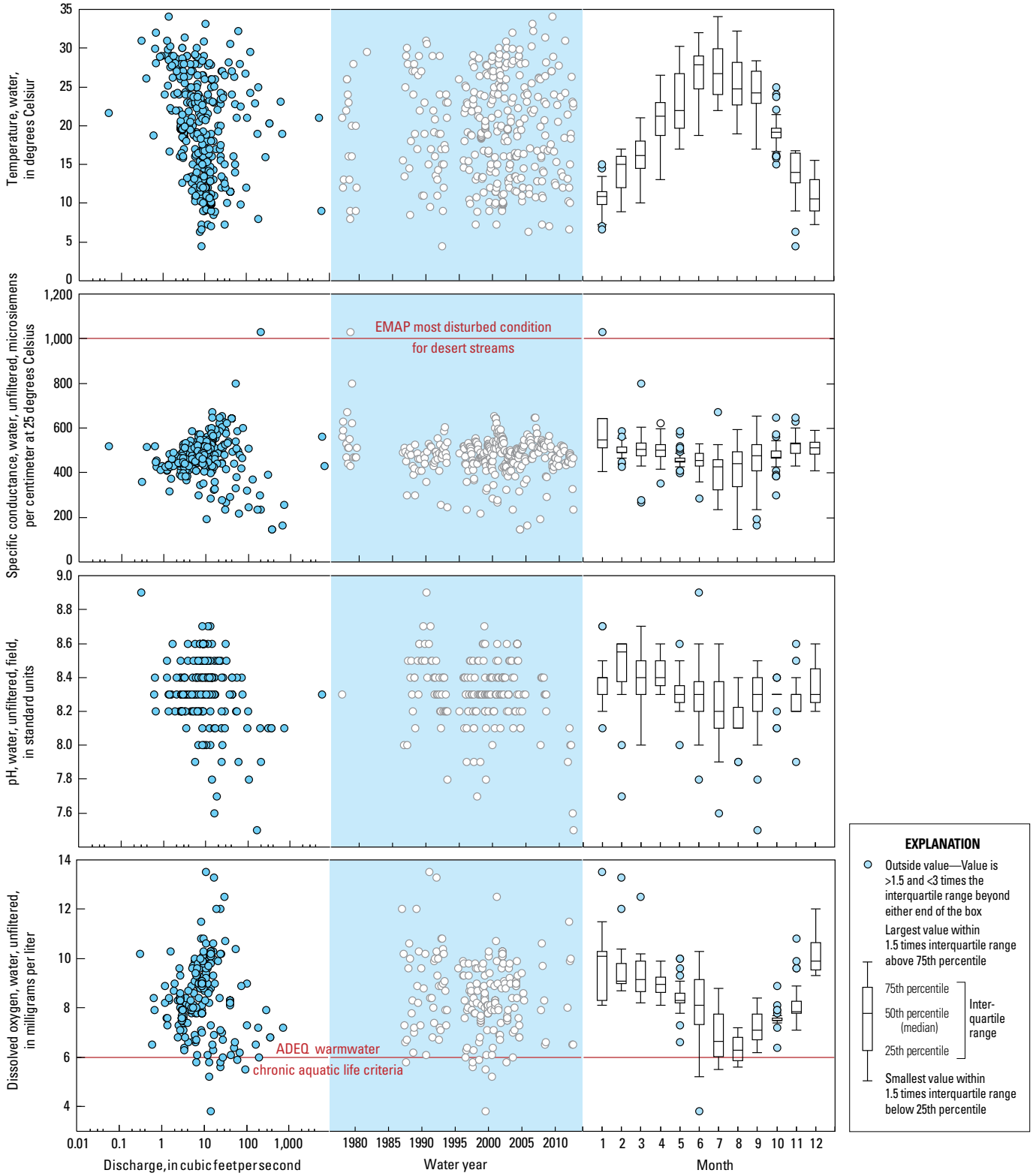


Figure 34. Plots of the distribution of water chemistry parameter measurements for water temperature, specific conductance, pH, and dissolved oxygen at the San Pedro River at Charleston gaging station in the Sierra Vista Subwatershed, southeastern Arizona. EMAP, Environmental Monitoring and Assessment Program ecological thresholds established by Arizona Game and Fish Department, U.S. Geological Survey, and U.S. Environmental Protection Agency; ADEQ, Arizona Department of Environmental Quality.

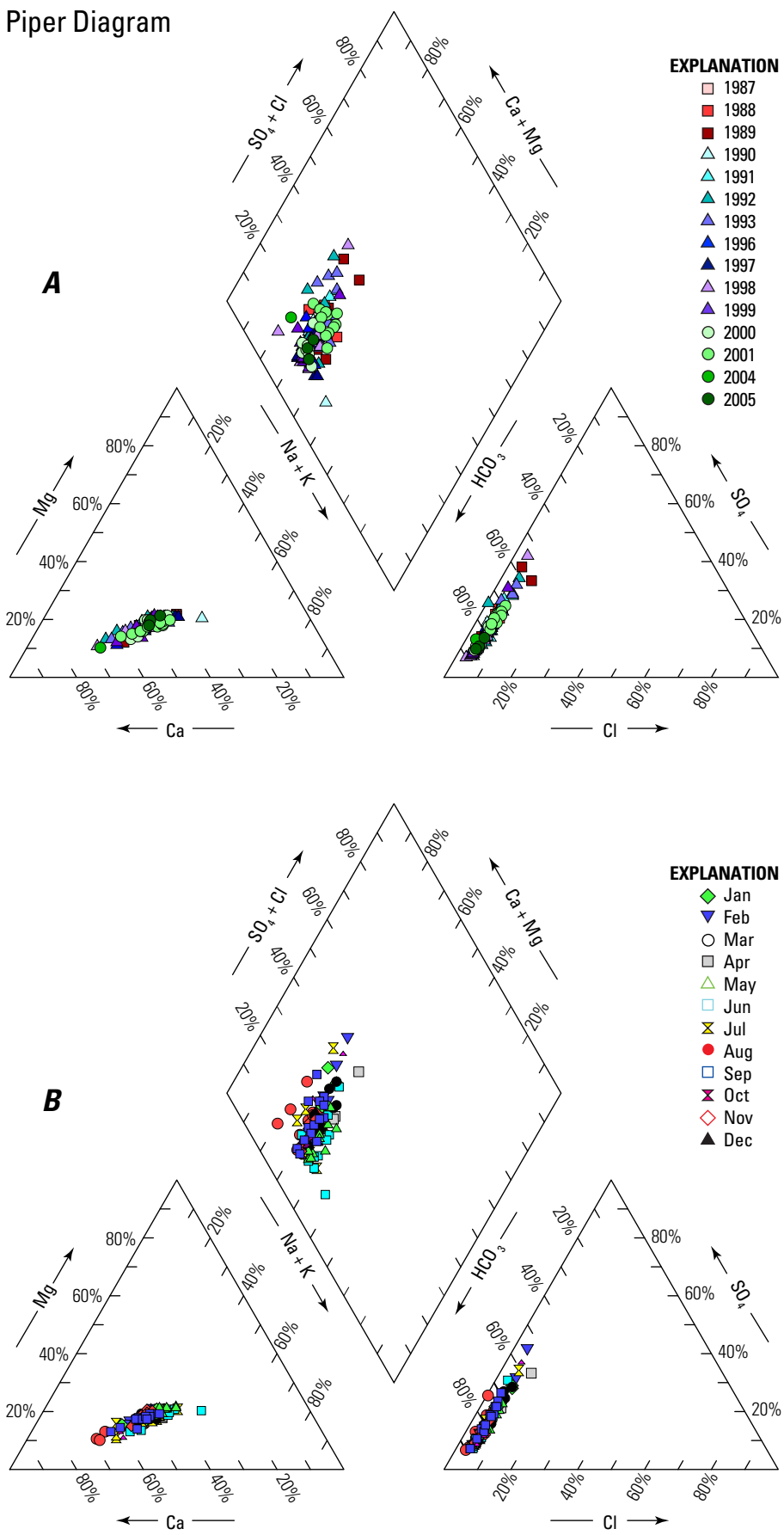


Figure 35. Piper diagrams showing the relative composition of major ions in surface water at the San Pedro River at Charleston gaging station in the Sierra Vista Subwatershed, southeastern Arizona. *A*, by year; *B*, by month. Ca, calcium; Cl, chlorine; Mg, magnesium; K, potassium; Na, sodium; SO_4, sulfate; HCO_3, bicarbonate; %, percent.

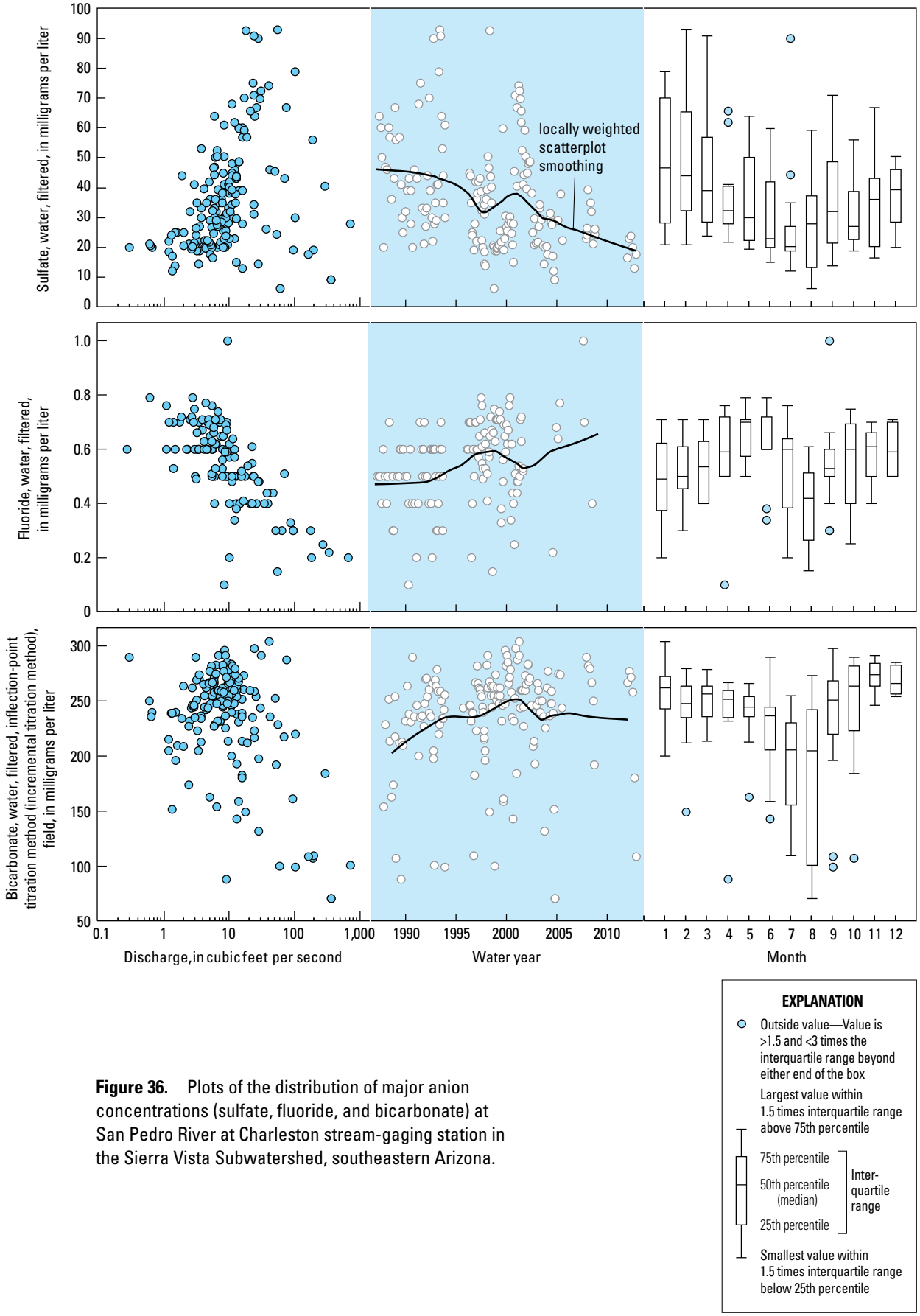


Figure 36. Plots of the distribution of major anion concentrations (sulfate, fluoride, and bicarbonate) at San Pedro River at Charleston stream-gaging station in the Sierra Vista Subwatershed, southeastern Arizona.

Nutrients

The ranges of nutrient concentrations observed in the San Pedro River were variable, but concentrations were also relatively low by comparison to eutrophic streams or levels in Arizona streams classified as impaired (Robinson and others, 2006). More than half of the total samples had no detected (or below detection limit) concentrations of total nitrogen, ammonia, and nitrate (table 4), and more than 25 percent of the concentrations were nondetects for total phosphorus and orthophosphate.

Five samples had concentrations of total nitrogen that exceeded the most disturbed condition (a sixth sample was censored) for the Arizona EMAP threshold for a desert (or xeric) stream (Robinson and others, 2006). Total nitrogen concentrations did increase with discharge, but the concentrations were somewhat variable as discharge increased (fig. 37). This was also reflected in the seasonal concentrations, where concentrations were greatest from June through August (the summer thunderstorm season), but concentrations were highly variable. Concentrations or exceedances did not appear to be increasing over time or with frequency during the period of analysis. Ammonia showed a

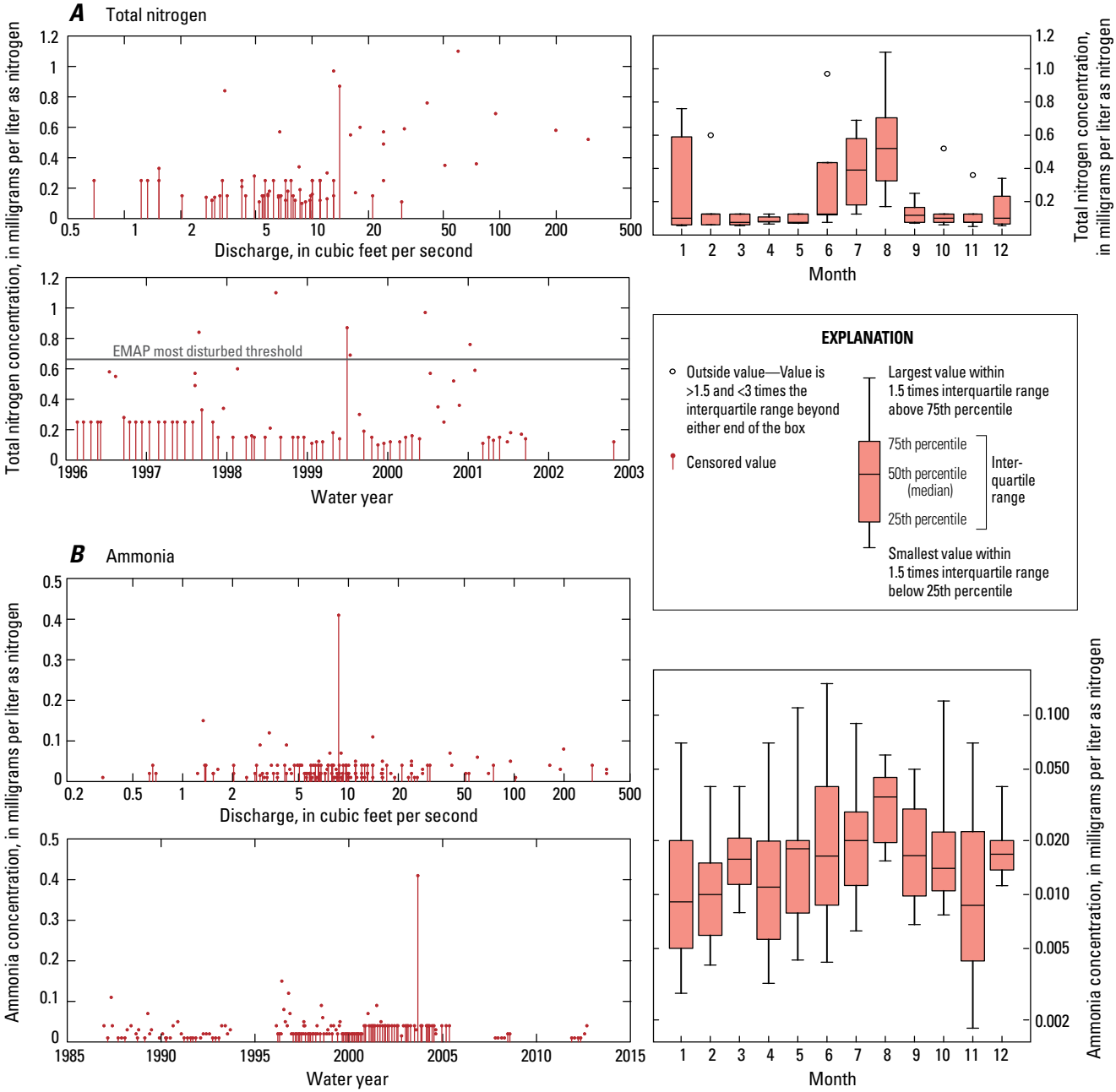


Figure 37. Plots showing (*A*) total nitrogen and (*B*) ammonia as a function of discharge and water year at the San Pedro River at Charleston gaging station in the Sierra Vista Subwatershed, southeastern Arizona. EMAP, Environmental Monitoring and Assessment Program ecological thresholds established by Arizona Game and Fish Department, U.S. Geological Survey, and U.S. Environmental Protection Agency.

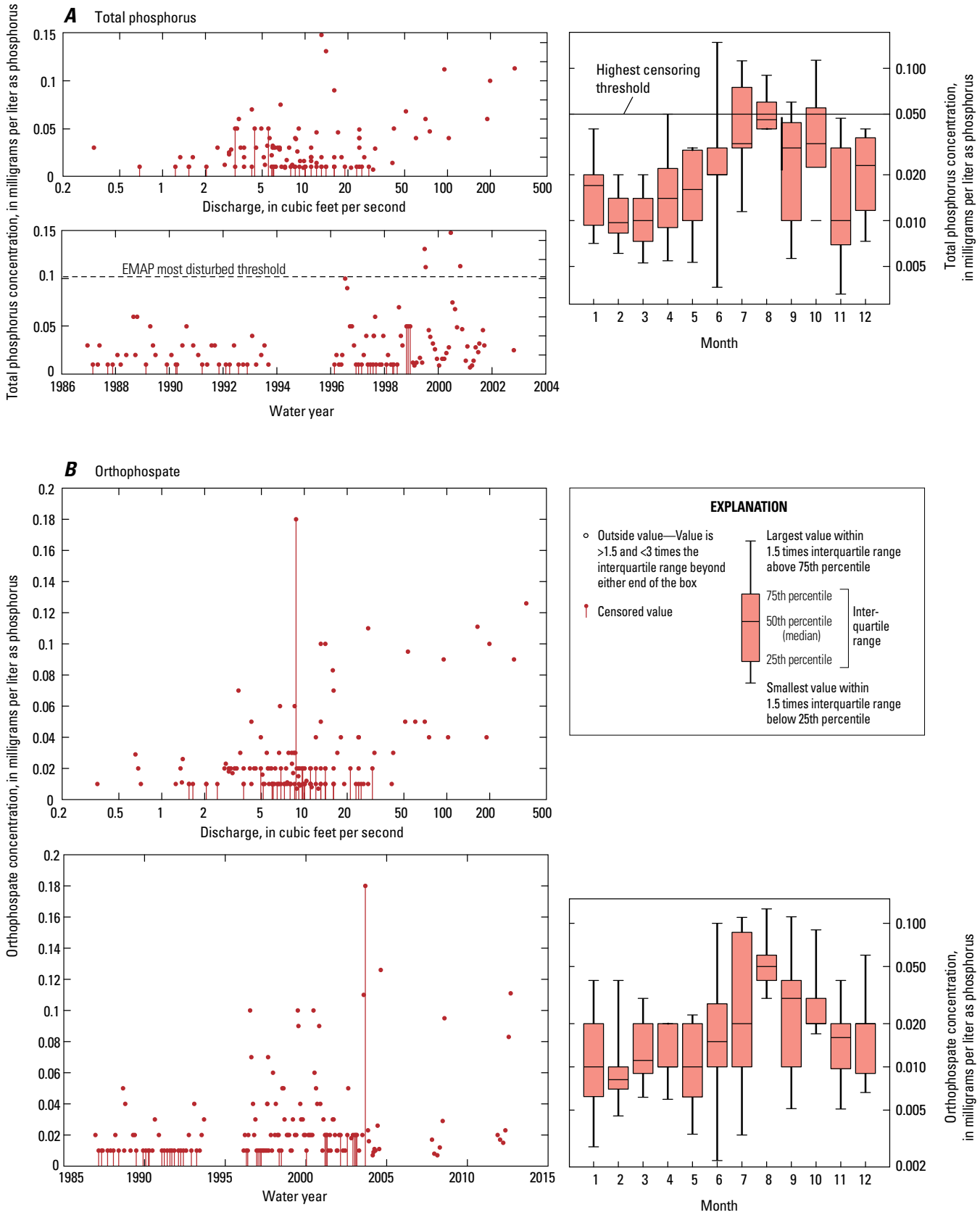


Figure 38. Plots showing (*A*) phosphate and (*B*) orthophosphate concentration as a function of discharge and water year at the San Pedro River at Charleston gaging station in the Sierra Vista Subwatershed, southeastern Arizona. EMAP, Environmental Monitoring and Assessment Program ecological thresholds established by Arizona Game and Fish Department, U.S. Geological Survey, and U.S. Environmental Protection Agency.

slight significant downward trend (τ=−0.121, p<0.0231), although sampling events were infrequent and the low-level detections were based on changing detection limits (table 4).

Total phosphorus concentrations exceeded the Arizona EMAP most disturbed ecological threshold four times. Total phosphorus trended upward over the period of analysis (τ=0.1282, p<0.0441; table 4). Orthophosphate or reactive phosphorus is the most stable kind of phosphate and is the form that is taken up by plants. Orthophosphate is produced by natural processes such as decomposition but can also be found in effluent from detergents and soaps. Orthophosphate concentrations showed a significant upward trend (τ=0.215, p<0.0001) during the period of analysis (table 4). Orthophosphate showed similar patterns to total nitrogen and phosphorus concentrations, both with concentrations increasing as discharge increased, and with maximum but variable concentrations occurring in the months July through September. Orthophosphate showed a more distinct increase with discharge than total phosphorus concentrations. The exceedances of total phosphorus and increase in both total phosphorus and orthophosphate over time indicate a potential upward trend at this stream reach.

Flow-normalized total phosphorus and orthophosphate concentrations showed a significant upward trend over the period of record, although the average flow-normalized concentrations were below the Arizona EMAP disturbed ecological threshold (fig. 38). The average change between 1988 and 2012 was about 0.049 mg/L and the mass flux was 5.5 t/yr. The late-1990s to the late-2000s showed the greatest increase in concentrations of about 7 to 8 percent per year. Orthophosphate showed a similar pattern but with more variability in recent years (fig. 38). The variability of the phosphorus indicators in recent years should be monitored in the future to determine if the upward trend continues.

Suspended Sediment

High levels of dissolved solids have a detrimental effect on aquatic biota, although there is no established aquatic-life criterion. Dissolved solids showed a significant upward trend based on the available data (τ, 0232; p<0.0001). Two data gaps (water year 1994–96 and 2001–04) and a few influential data points make it difficult to conclude that this trend will necessarily continue. Similar to specific conductance, there is a slope change (negative) related to increased discharge or rainfall runoff events. Spring runoff months have the greatest concentrations of dissolved solids, whereas July and August concentrations were the lowest. Dissolved solids concentrations get diluted from stormwater runoff during the convective storms in July and August in locations such as the subwatershed, where the dominant land use is neither agricultural nor urban (Cordy and others, 2000).

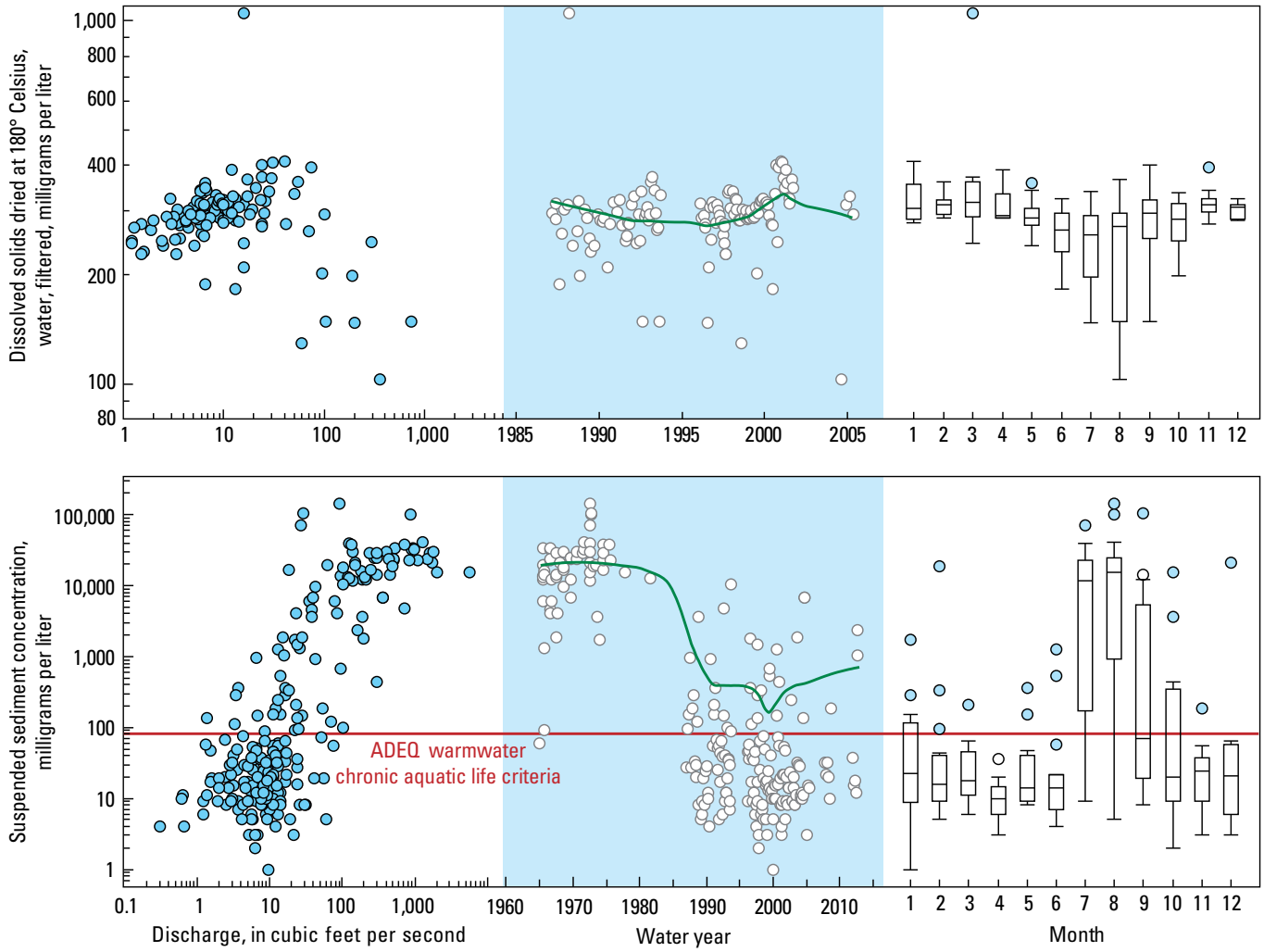
Concentrations of suspended sediment at the Charleston location span several orders of magnitude over the period of analysis (>100,000 mg/L and <2 mg/L; fig. 39). For stream flows of less than 10 ft^3/s, suspended-sediment concentrations were generally stable (between 1 and 100 mg/L), but for flows of 10 ft^3/s and above, concentrations varied more widely and increased markedly with increasing discharge. The months with highest discharge (July, August, and September) had the greatest and most variable (four orders of magnitude) suspended-sediment concentrations during these months (fig. 39).

The median concentration of suspended sediment was 48 mg/L. The ADEQ warm-water aquatic criterion is 80 mg/L, which is frequently exceeded during floods. The number of exceedances has not increased since routine sampling was resumed around 1985. One of the strongest downward trends occurred in the suspended sediment concentrations between mid-1960s and late 2013 (τ, −0.438; p<0.0001). There is a visible step-change in the time-series plot of discharge versus suspended sediment concentration between 1970 and 1980 (fig. 39). This marked shift is also evident in the flow-normalized results (fig. 40).

The flow-normalized suspended sediment concentrations decreased on average 1,860 mg/L between1966 and 2012. This is equivalent to about 10,200 t/yr, although there was continual variability in the concentrations (fig. 40). The period of analysis shows a steady and slightly upward trend between 1966 and 1971. The greatest downward slope of the average concentrations occurred between late 1970s and early 1980s, when average concentrations decreased about 1,800 mg/L (50–60 percent change). Between 1980 and 1990, concentrations continued to decrease, but more gradually than the previous downward trending period. The period of record between 1990 and 2000 had the lowest average suspended sediment concentrations for the period of record. After 2000, the concentrations become more variable, and there were sustained periods of high and low concentrations. The overall pattern of this period was a gradual upward trend.

Decreases in suspended sediment at Charleston from 1966 to the mid-1980s may be related to changes in land-use practices in the subwatershed, as well as to changes in the density and extent of the riparian forest. The total number of cattle in the subwatershed peaked in the late 1970s and declined gradually until 1986 when their numbers dropped by about 50 percent. The cattle population continued to decline by about 10 percent per year from 1987 to 2008, at which time it stabilized (James Lindsey, Chairman, Hereford Natural Resource Conservation District, written commun., 2013). Cattle were also removed from all but 6,521 acres of the SPRNCA on its designation in 1988 (Bureau of Land Management, 1991), ending much of their access to the flood plain and river. The one major exception to this is the Brunchow Hill grazing allotment, which traverses the riparian forest and river immediately upstream of the Charleston gaging station (Bureau of Land Management, 2011). Flow-normalized suspended sediment at Charleston also dropped from around 2,500 mg/L to 1,000 mg/L to 500 mg/L in the 1970s, 1980s, and 1990s, respectively, before rebounding somewhat in subsequent years.

A second factor that may have played a role in the reduction of suspended sediment measured at Charleston was cottonwood and willow recruitment along much of the river length. Cottonwood cohorts established a large area along the flood plain in the 1960s, with decreases in total recruitment and



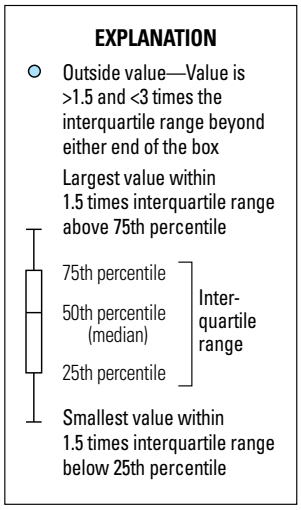


Figure 39. Plots showing dissolved solids and suspended sediment concentration at the San Pedro River at Charleston gaging station in the Sierra Vista Subwatershed, southeastern Arizona. ADEQ, Arizona Department of Environmental Quality.

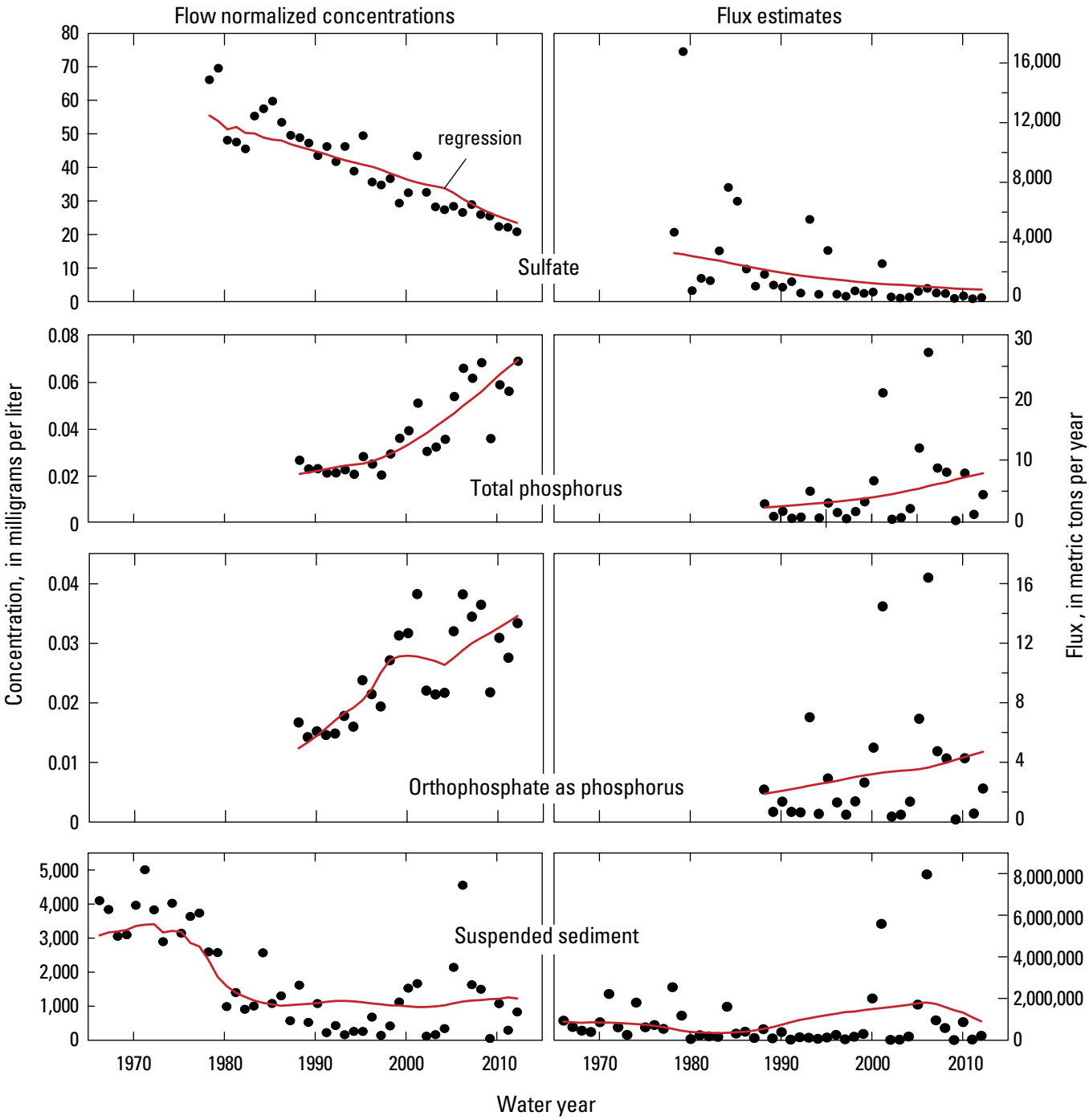


Figure 40. Plots showing flow-normalized concentrations (left, points), flux estimate (right, points), and regressions for flow normalized concentrations and fluxes (lines) for sulfate, total phosphorus, orthophosphate, and suspended sediment at the San Pedro River at Charleston gaging station in the Sierra Vista Subwatershed, southeastern Arizona.

additional coverage in subsequent decades (Stromberg and others, 2010). Plant growth increases hydraulic roughness that reduces flow and flood peaks and in turn facilitates sediment deposition (Huckleberry and others, 2009). Removal of most cattle from the SPRNCA may have facilitated this process as well (Huckleberry, 1996).

A third factor that may have been at least in part responsible for the reduction of suspended sediment measured at Charleston was the curtailment of sand-and-gravel operations from the SPRNCA with its designation in 1988. In particular, a sand-and-gravel operation 7 miles upstream of Charleston and adjacent to the San Pedro River could have been a major contributor to the sediment flux at Charleston before 1988 (B. Lomeli, BLM Hydrologist, oral commun., 2015).

The sediment flux showed a different trend (fig. 40). Flux does not necessarily track with concentration. Concentration can be high, but if discharge is low, flux will remain relatively low. Unlike suspended-sediment concentrations, which were high in the early part of the record, the flux was relatively low, and overall, the trend was downward into the 1980s and somewhat similar to the concentration pattern. The 1990s were marked by the consistently lowest suspended-sediment concentrations in the period of record, although an upward trend emerged toward the end of the decade. The upward trend lasted to 2006, a year of sustained monsoon flooding. Debris flows in the Huachuca Mountains in 2006 mobilized large volumes of sediment from 113 slope failures (Webb and others, 2008), with some of that sediment likely transported to the San Pedro River during that very wet summer. After 2006, summer precipitation declined and the pattern trended downward.

Trace Elements

Cadmium, chromium, lead, nickel, selenium, and zinc concentrations were mostly not detectable (nondetections for the analyses applied) during the period of collection. Copper and arsenic concentrations had more detections than most other elements analyzed, but concentrations were still below the EPA's aquatic-life and drinking-water criteria (table 4 and fig. 41). Arsenic and copper showed significant trends over the period of analysis (τ, 0.198; p=0.0187 and τ, −0.373; p<0.0001, respectively). Copper concentrations decreased significantly from the late 1980s to the early 1990s following several tailings dam failures between December 1977 and April 1985 at mines

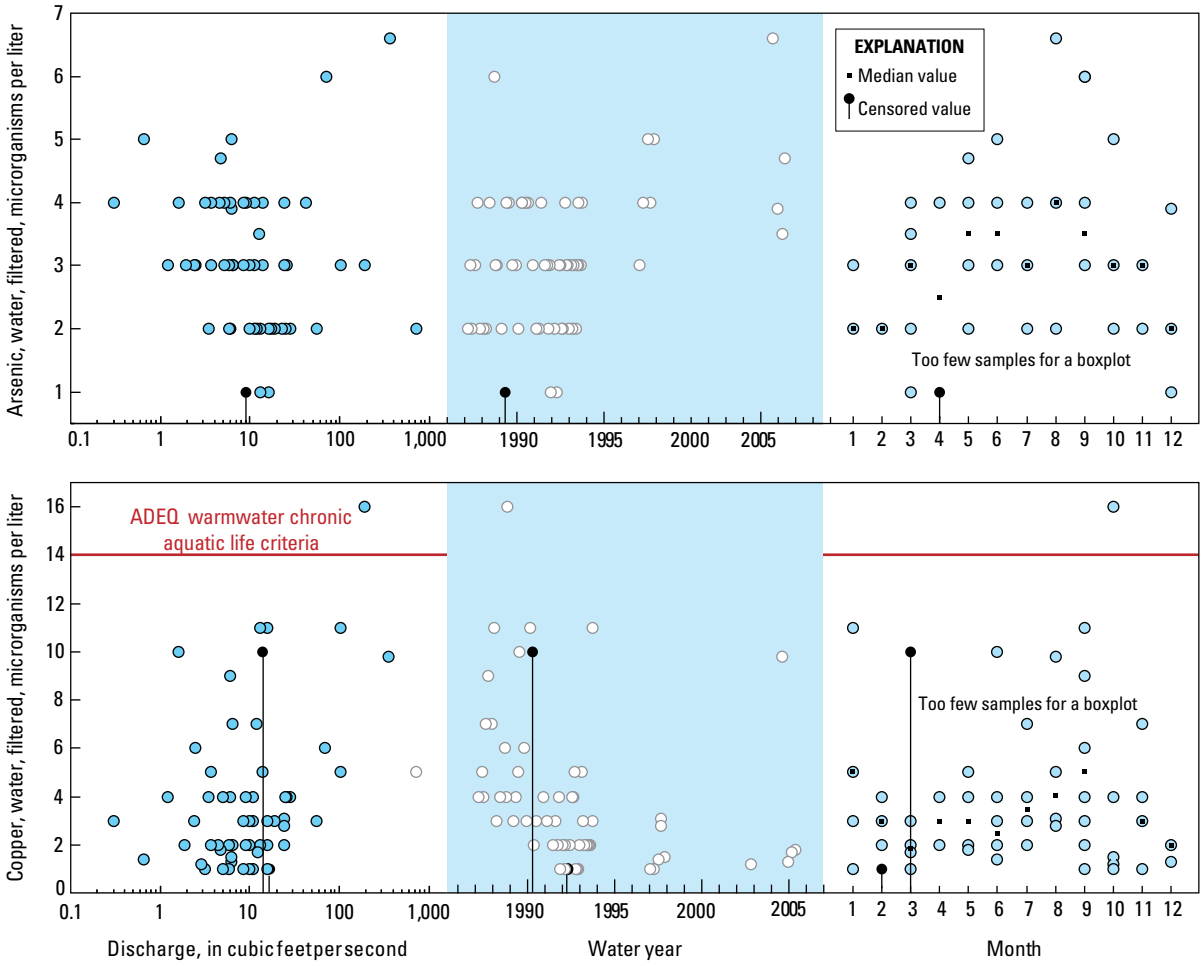


Figure 41. Plots showing arsenic and copper concentrations at the San Pedro River at Charleston gaging station in the Sierra Vista Subwatershed, southeastern Arizona. ADEQ, Arizona Department of Environmental Quality.

in Cananea, Sonora, Mexico, located near the headwaters of the San Pedro River (King and others, 1992). Median copper concentrations were highest in January, August, and September but were still several orders of magnitude lower than the drinking water standard (fig. 41). All arsenic concentrations analyzed from Charleston were below the drinking-water standard. Monthly concentrations of arsenic were highest in the summer months and overall more variable than most constituents.

Indicator 12. Isotope Analysis—Methods

To investigate groundwater inputs to river flow and variability through time, the stable-isotope ratios of oxygen ($\delta^{18}O$) and hydrogen ($\delta^{2}H$) were analyzed. Comparison of the isotopic signature of samples from the San Pedro River with the signature in regional-aquifer wells can help to evaluate trends in water source through time and to evaluate whether source water in the San Pedro River is changing. Such a change would confirm model characterizations that show groundwater pumping has begun to capture San Pedro River surface flow (Pool and Dickinson, 2007; Lacher, 2012; Lacher and others, 2014), and if so, where.

Surface water and almost all groundwater originate as precipitation. Values of $\delta^{18}O$ and $\delta^{2}H$ in precipitation vary as a function of source vapor composition and with condensation temperature. As a result, $\delta^{18}O$ and $\delta^{2}H$ values in the rain of a particular area may vary with season and with the altitude of precipitation. In general, higher elevations receive more isotopically depleted rainfall (lower values of $\delta^{18}O$ and $\delta^{2}H$). This effect occurs even where topographic relief is minor (Clark and Fritz, 1997) and results in variations in groundwater isotopic composition in the subwatershed that correlate with recharge source-area elevation (Pool and Coes, 1999). Once precipitation leaves the atmosphere, runs off, and becomes recharge, the isotopic signature of the water generally does not change except through evaporation (in which heavier isotopes are left in the liquid phase resulting in higher isotope ratios) or from mixing with waters of different isotopic content (Kendall and others, 2004). On a plot of $\delta^{18}O$ and $\delta^{2}H$, residual water left after partial evaporation of surface water typically produces data trends with slopes near 4 or 5 at lower humidity (Clark and Fritz, 1997). In the Southwestern United States, published examples include Rio Grande surface water with a slope of 5.1 (Eastoe and others, 2008), and lower Colorado River surface water with slopes of 5.1 and 5.6 (Guay and others, 2006).

In southern Arizona, multiyear datasets of $\delta^{18}O$ and $\delta^{2}H$ values for individual rain events were available for Tucson and the Palisades Ranger Station in the Santa Catalina Mountains (Eastoe and others, 2004; Wright, 2001), where the climate and altitude effects were considered similar to those of the present study area. Eastoe and Dettman (2016) found that isotope values for the individual events vary greatly, regardless of season, and defined two linear trends rather than a single local meteoric water line (fig. 42). For both summer and winter data with $\delta^{18}O$ values <−6 per mil, the trend has a slope of 7.8 and an intercept of 9.9 and is indistinguishable from the global meteoric water line (GMWL). For data with $\delta^{18}O$ values >−6 per mil, the trend has a slope of 5.0 and an intercept of 7.0 and is an evaporation trend; summer rain events were the main contributors. The seasonal long-term amount-weighted averages were distinctive ($\delta^{18}O$, $\delta^{2}H$) being (−6.0, −42) for summer and (−8.9, −59) for winter at the elevation of central Tucson. Amount-weighted seasonal averages vary with altitude (Eastoe and others, 2004). Smaller datasets for precipitation in the Sierra Vista Subwatershed (Baillie and others, 2007; Coes and Pool, 2007) indicate general agreement with the isotope character of precipitation in the Tucson Basin but were probably too limited to yield precise averages.

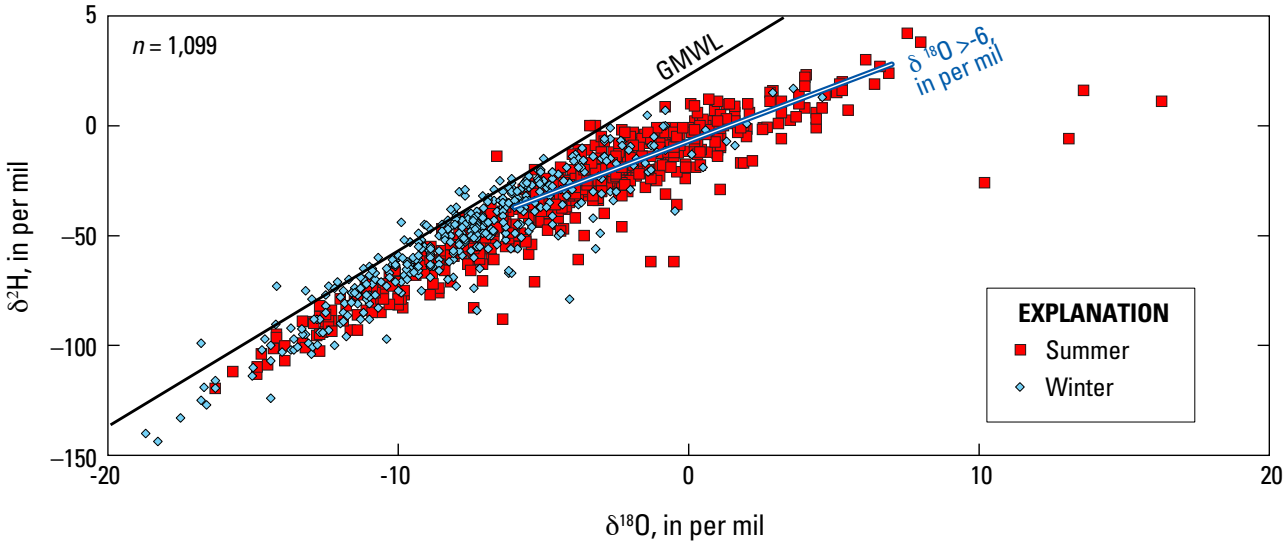


Figure 42. Graph showing oxygen ($\delta^{18}O$) and hydrogen ($\delta^{2}H$) isotope ratios in samples of precipitation from Tucson, Arizona, plotted in relation to the local meteoric water line (Eastoe and Dettman, 2016). Isotopic-ratio values are shown relative to Vienna Standard Mean Ocean Water. Data collected near the University of Arizona at about 2,400 ft above sea level. Summer is June–October; winter is November–May. GMWL, global meteoric water line. > greater than.

Surface-water samples for isotope measurement were consistently collected about monthly since the middle to late 1990s at the three mainstem gaging stations (Palominas (reach 1), Charleston (reach 7), and Tombstone (reach 11)), as well as at the Hereford bridge (reach 2), the Lewis Springs stage recorder (reach 5), and the Lower Babocomari gaging station (reach 9). Samples were collected and processed according to standard protocols (U.S. Geological Survey, variously dated). Analyses were performed at the USGS Stable Isotope Laboratory in Reston, Virginia, using standard methods (Révész and others, 2008a and 2008b). Analytical precisions (2σ) were 2 per mil for δ^2H and 0.2 per mil for $\delta^{18}O$. Isotope samples have also been collected by various researchers at regional-aquifer wells, alluvial-aquifer wells, and springs across the subwatershed (Pool and Coes, 1999; Goodrich and others, 2004; Baillie and others, 2007; Wahi and others, 2008).

Trends at the six sampling locations were analyzed to understand the processes controlling seasonal and long-term variations in isotope ratios. Seasonal variations can be related to seasonal differences in the isotope composition of precipitation (Eastoe and Dettman, 2016) and evaporation (as affected by windspeed, humidity, and temperature). In this region, long-term trends at the timescale of sampling appear to be unrelated to changes in precipitation volume (Eastoe and Dettman, 2016) but may be caused by changes in groundwater and surface-water flow regimes. The data analysis to be presented here depends on spatial changes (from reach to reach of the San Pedro River) in the slopes of long-term data trends. Evaporation trends (slopes of 4 to 5 as mentioned above) were expected to be consistent across the study area, and any departure from the local evaporation slope can be interpreted in terms of a combination of water mixing with evaporation (Clark and Fritz, 1997).

Indicator 12. Isotope Analysis—Results

The $\delta^{18}O$ data can be resolved into two patterns that were evident in the record at all sites—cyclic variations with amplitudes of about 1.5 per mil or less that correspond to times of low flow and sporadic outlying $\delta^{18}O$ values that correspond to times of high flow in response to precipitation events. The cyclic variations in the time-series data (fig. 43) might result from seasonal evaporation or from changes in the mixing ratio of groundwater and surface water, as the availability of the latter changes with season. The outlying values may be higher or lower than the typical limits of the cyclic variations and usually correspond to precipitation events. Outliers occur most frequently during the summer monsoon and affect $\delta^{18}O$ values in subsequent river discharge during the summer and fall. At Palominas, lower $\delta^{18}O$ values typically occur in the winter months. Higher $\delta^{18}O$ values often occur in June or early July. Similar observations apply at the Lower Babocomari and Tombstone gage sites. Values of $\delta^{18}O$ at Hereford, Lewis Springs, and Charleston were roughly similar; they increase through the winter and decrease in late spring. Statistically significant trends over the period of record (using the Mann-Kendall test; Salmi and others, 2002) in median annual values of $\delta^{18}O$ for the period December to June were found at Lewis Springs (p=0.05), the Babocomari River Near Tombstone gaging station (p=0.05), and Palominas (p=0.01). All trends in these reaches were positive; that is, $\delta^{18}O$ values were increasing over the period of record. The positive trend could reflect a decrease in the amount of groundwater entering the river channel as base flow at these locations. At Lewis Springs, another plausible explanation is that beaver ponds, which were established in the early 2000s, could provide a source of evaporated water to the river and so contribute to the increasing $\delta^{18}O$ value.

Figure 44 shows all $\delta^{18}O$ and δ^2H data for July to November (when most high flows occur) grouped by upstream and downstream surface-water sites. The isotope data for large discharge events that plot outside the diffuse central clusters of data mainly fall close to the GMWL, indicating that the water corresponding to these events has undergone little evaporation. The outliers in the July–November values include events with $\delta^{18}O$ and δ^2H values both higher and lower than those of the central clusters, consistent with the broad range of isotope composition of "summer" (June to October) rain in Tucson (Eastoe and Dettman, 2016). The outliers represent chance coincidence of sampling with a particular rainfall-induced runoff event at a given gage, rather than a systematic sampling of all such events.

The December to June data in figure 45*A–D* are presented by mainstem reach rather than grouped together as they were in figure 42, so that isotopic and hydrologic trends and processes can be analyzed reach by reach. Base-flow conditions prevail during these months, with a narrow range of $\delta^{18}O$ (mainly between −6 and −8 per mil; fig. 45 *A–D*) reflecting contributions of groundwater. Typically, few large flow events and isotope outliers were present during this part of the year. The cluster of data for each location makes up an elongate array with a long axis of positive slope, ranging from 3.7 to 5.9. Outliers have been omitted because they probably represent rainfall-runoff events and not base flow.

The following features of the data were significant for the interpretation of the graphs in figure 45:

1. The tight clustering of ($\delta^{18}O$, δ^2H) data (outliers excluded) between December and June, when base flow conditions prevail;
2. The occurrence of higher $\delta^{18}O$ values (outliers excluded) in June and early July—commonly the hottest, driest part of the year—at the Palominas, Lower Babocomari and Tombstone gages, pointing to evaporation as the principal control of isotope variation within the data clusters in the reaches immediately upstream of these sites;
3. The decrease in $\delta^{18}O$ values (outliers excluded) in late spring at the other three sites, pointing to an additional process in the reaches immediately upstream of those sites; and
4. The differences in slopes of the data clusters in figures 45*A*, *B*, *C*, and *D* also indicate that more than one process is responsible for isotope variation in base flow.

Isotope data from earlier publications were relevant to this interpretation. In the Sierra Vista Subwatershed, the isotope

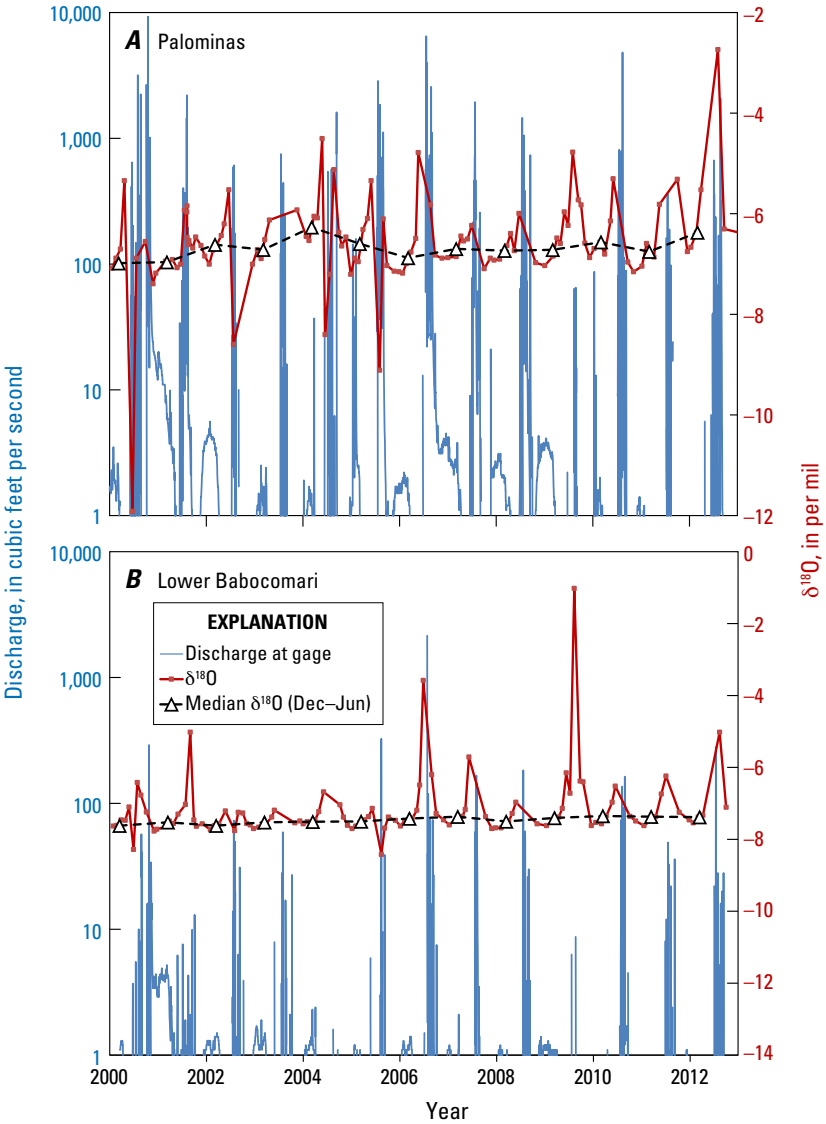


Figure 43. Graphs showing oxygen ($\delta^{18}O$; all data and annual median) plotted for the (*A*) Palominas and (*B*) Babocomari near Tombstone sampling locations and stream discharge at nearest stream gage in the Sierra Vista Subwatershed, southeastern Arizona. Both sites show a statistically significant increasing trend over the period of record.

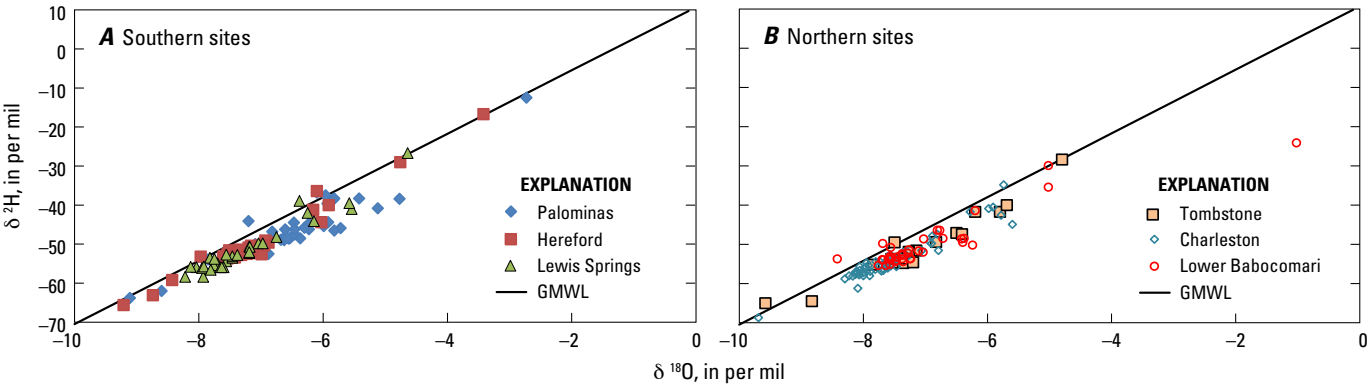


Figure 44. Graphs showing San Pedro River and Babocomari River oxygen ($\delta^{18}O$) and hydrogen ($\delta^{2}H$) data in the Sierra Vista Subwatershed, southeastern Arizona, for July to November plotted with the global meteoric water line (GMWL). *A*, Southern, upstream sites, and, *B*, northern, downstream sites.

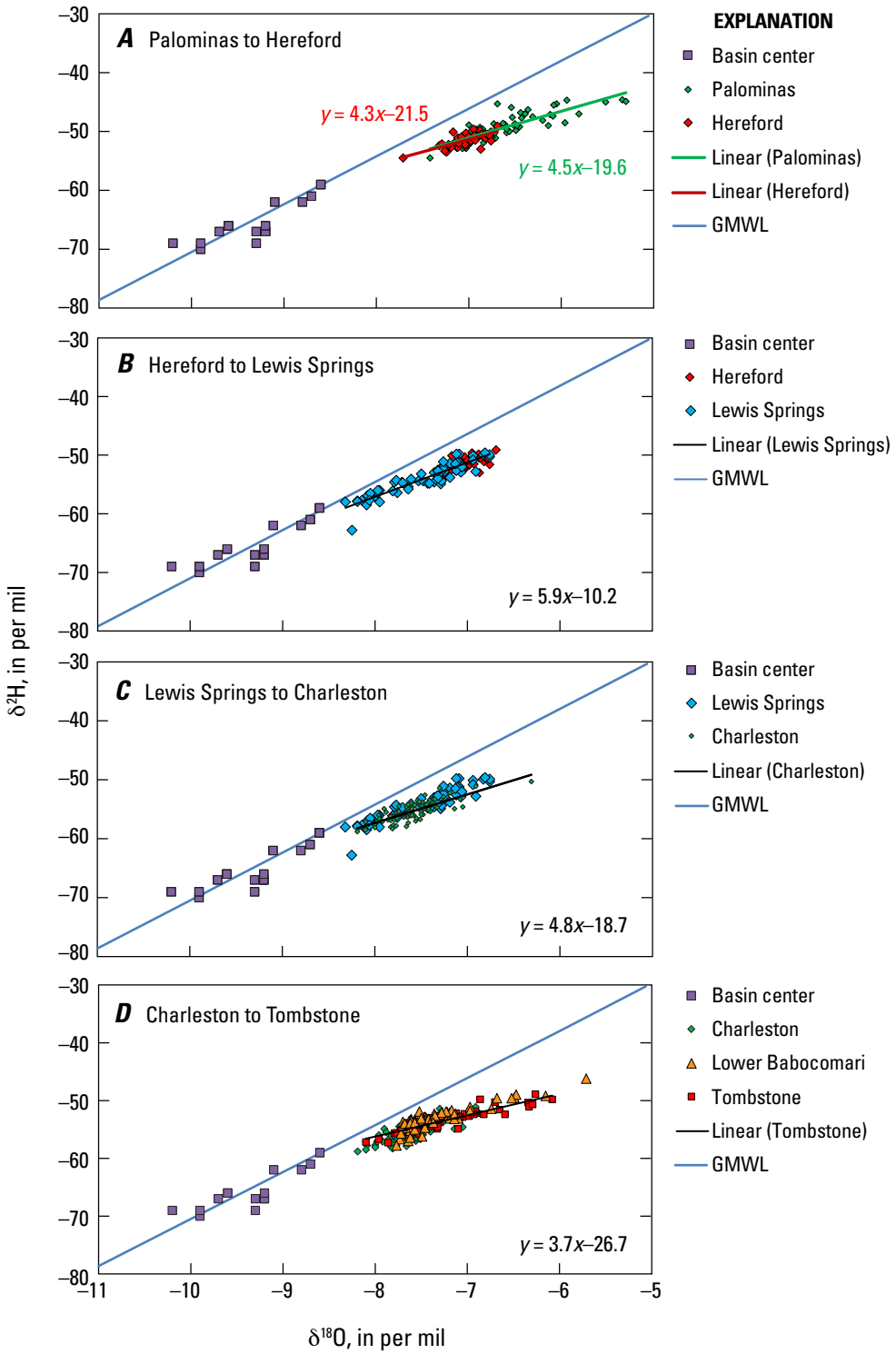


Figure 45. Graphs showing basin-center and San Pedro River base-flow plots and slopes of oxygen ($\delta^{18}O$) versus hydrogen ($\delta^{2}H$) for December to June (with outliers removed) in the Sierra Vista Subwatershed, Upper San Pedro Basin, southeastern Arizona. Mixing of basin-center groundwater may begin at Palominas (*A*, slope 4.3), but the groundwater contribution clearly peaks from Hereford (slope 4.5) to Lewis Springs (*B*) where the slope is greatest (5.9). Base-flow isotopic slope at Charleston (*C*, 4.8) overlaps that at Lewis Springs, indicating a smaller basin groundwater contribution downstream of Lewis Springs. The change in slope of the data array from Lewis Springs to Charleston is consistent with a transition to mixing of basin groundwater along with some evaporation as concurrent controls on the isotopic variation. There is little or no mixing of basin-center groundwater into the river from Charleston to Tombstone (*D*, slope 3.7), and evaporation controls the variation of isotopes in base flow. The water that arrives at the Tombstone gage can be interpreted as an evaporated mixture of water from Charleston and water from the Babocomari River (not shown).

altitude effect of the Huachuca Mountains presents low $\delta^{18}O$ and $\delta^{2}H$ values of mountain spring water and a zone of groundwater, largely mountain-derived, also with low $\delta^{18}O$ and $\delta^{2}H$ values relative to other groundwater in the basin alluvium (Pool and Coes, 1999; Wahi and others, 2008; Kennedy and Gungle, 2010). The $\delta^{18}O$ and $\delta^{2}H$ data for the basin-alluvium groundwater from Wahi and others (2008) ("basin center") were compared with river base-flow data in figs. 45 *A–D*. Pool and Coes (1999), Baillie and others (2007), and Kennedy and Gungle (2010) proposed that the discharge of groundwater from the alluvial basin south of Sierra Vista to the river and the stream alluvium is likely to affect the isotopic composition of the river water. In a similar way, the time-series isotope data presented here for base flow and flood stages in the river, in combination with river discharge data, can potentially help to differentiate the relative contributions to the river of discharge from summer bank storage and from the regional aquifer.

Values of $\delta^{18}O$ and $\delta^{2}H$ declined from Palominas to Hereford (fig. 45*A*). Isotope values at Hereford plot between values of basin groundwater and values at Palominas, indicating potential input of groundwater in the reach between Hereford and Palominas. The Lewis Springs data (fig. 45*B*) plot on a straight line between base flow at Hereford and basin groundwater, consistent with mixing between these two end-members as the explanation of the data array at Lewis Springs. The slope of the Lewis Springs trend (5.9) is greater that the slopes in fig. 45*A*. Similar considerations apply at Charleston, where the data array overlaps that at Lewis Springs but has a lower slope (4.8); nonetheless, the trend remains consistent with mixing of basin groundwater with Lewis Springs base flow. The lower slope may reflect a growing influence of evaporation between Lewis Springs and Charleston.

The data for the Tombstone gage, where the slope is 3.7 (fig. 45*D*), offer the clearest evidence of a reach of the river where evaporation is the main cause of isotope variation. The Tombstone gage lies downstream of the confluence of the Babocomari and San Pedro Rivers. Values of $\delta^{18}O$ and $\delta^{2}H$ for the Lower Babocomari gage were slightly higher on average than those for the Charleston gage (fig. 45*D*); a mixture of the two provides water of an isotope composition that evolves by evaporation (as suggested above on the basis of seasonal variation in $\delta^{18}O$) between the confluence of the Babocomari and San Pedro Rivers and the Tombstone gage. The linear trend for the Tombstone gage data has the lowest of the slopes in fig. 45 and is the most likely to represent evaporation without groundwater mixing.

The combination of evaporation and other effects to yield higher slopes can be further demonstrated for Palominas and Lower Babocomari by separating data for December to March and April to June (fig. 46). April to June is the part of the year expected to have the highest evaporation rates. At Palominas, two trends emerge—(1) an evaporation trend of slope 3.6 for April to June and (2) a trend with slope 6.2 for December to March. The

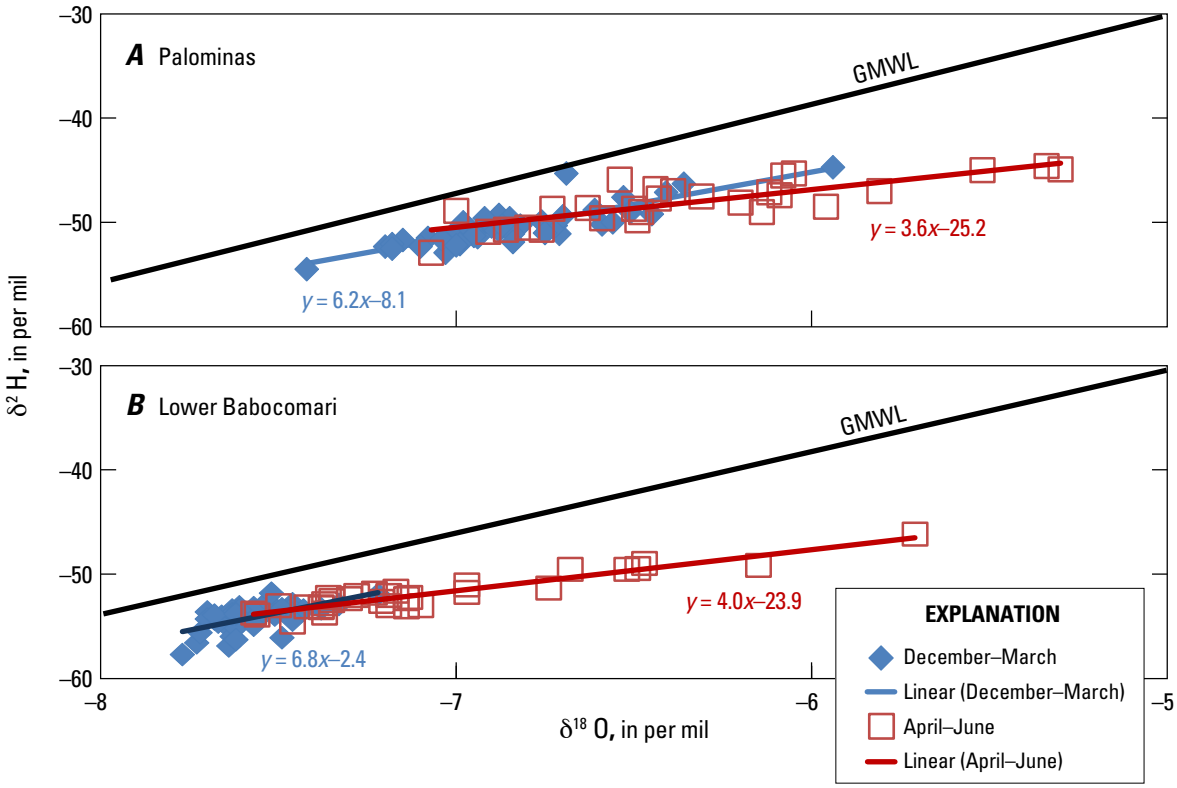


Figure 46. Graphs showing San Pedro River base-flow plots and slopes of hydrogen ($\delta^{2}H$) versus oxygen ($\delta^{18}O$) for December to March and April to June at two gaging station locations in the Sierra Vista Subwatershed, southeastern Arizona. *A*, Palominas; *B*, Lower Babocomari. GMWL, global meteoric water line.

latter may be a mixing trend indicating groundwater discharge into the river reach upstream of the gage as proposed by Kennedy and Gungle (2010); however, no data for likely groundwater end-members in that area were available. At the Lower Babocomari gage, an evaporation trend of slope 4.0 is present in April to June. What is shown on figure 46*B* as a trend line of slope 6.8 for December to March is based on few data and is therefore likely to be imprecise. Data from the other four locations show no such clear separation of trends.

The long-term isotope data from the six sites indicate that the main groundwater contributions to the San Pedro River occur between the Hereford bridge and the Charleston gage. A smaller contribution may occur between the Palominas gage and the Hereford bridge. The slopes of the isotope-data trends for the reaches immediately upstream of the Palominas and Tombstone gages were controlled mainly by evaporation. Combining this understanding of isotope variation with the long-term increase in $\delta^{18}O$ at the Palominas gage, where there is evidence of groundwater discharge into the river, suggests a decrease over the period of observation in the groundwater contribution to base flow in these reaches.

Group 4. Springs Indicators

Indicator 13. Springs Discharge—Methods

The USGS measured discharge from four springs and one artesian well quarterly in the Sierra Vista Subwatershed (fig. 47). Springs with higher discharge (Murray and Lewis Springs) were measured with a current meter, springs with lower discharge (Horsethief Spring and Moson Spring) were measured with a flume, and the McDowell-Craig Farm artesian well was measured volumetrically. Murray Springs discharge was measured quarterly from 2003 to 2012. Horsethief Spring, Lewis Springs, and the McDowell-Craig Farm well were measured quarterly from 2005 to 2012. Moson Spring was first measured in 2006 and then was measured quarterly from the last quarter of 2008 to the last quarter of 2012. The Mann-Kendall tau statistic, which is well suited for measuring the strength of a monotonic trend between two variables that exhibit skewness around the general relation, was used to assess spring discharge trends (Helsel and Hirsch, 2002; Macleod, 2011).

Indicator 13. Springs Discharge—Results

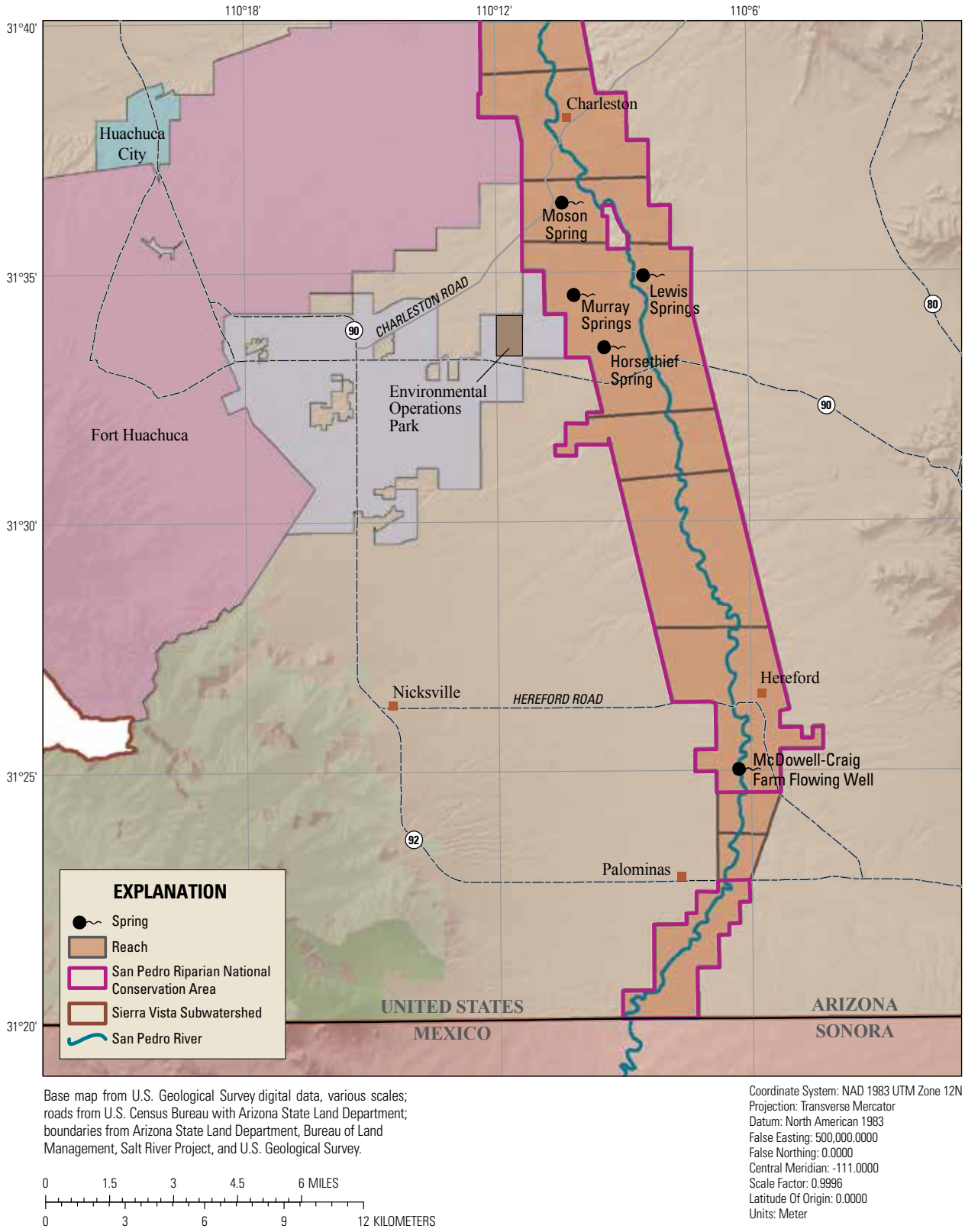
Median annual discharge measured at the five sites ranged from less than 0.01 ft^3/s (Horsethief Spring in 2005) to about 0.48 ft^3/s (Murray Springs in 2009) (fig. 48; note actual discharge at Horsethief Spring and Moson Spring was one-tenth the values shown on fig. 48*A* left vertical axis). Murray Springs had the greatest range of annual medians among the five sites, from 0.01 ft^3/s in 2003 to 0.48 ft^3/s in 2009. Horsethief Spring did not flow in late spring/early summer of 2005 and 2006, and McDowell-Craig Well did not flow in late spring/early summer of 2006 and 2012. Springs on the west side of the San Pedro River—Horsethief, Murray, and Moson—showed a similar, mostly increasing trend in discharge until 2009, when discharge began to decrease (fig. 48*A*). The City of Sierra Vista's EOP recharges treated effluent into a perched zone above a silt and clay layer up-gradient of Murray Springs. This shallow recharge began in 2003 and likely led to the increase in Murray Springs discharge beginning in about 2005 (Brown and Caldwell, 2009). By 2009, Curry Draw flowed from Murray Springs to within at least a quarter mile of the San Pedro River even during the driest time of the year (The Nature Conservancy, 2015). Horsethief Spring and Moson Spring discharge had a similar but less pronounced pattern to that of Murray Springs discharge and may have experienced a similar but reduced influence from the EOP recharge (fig. 48*A*).

McDowell-Craig Farm flowing well and Lewis Springs, both east of the river but 11 miles apart south to north, have similar magnitudes of flow, although this is coincidental (fig. 48*B*). The general pattern of discharge at the east-side springs is similar to the annual precipitation pattern, whereas the overall decline in spring discharge from 2008 to 2012 may be related to a reduction in winter precipitation that began around 2001(fig. 48*B*).

Indicator 14. Springs Water Quality—Methods

Water-quality analyses, including wastewater and pharmaceutical suites, were completed four times in Curry Draw and once each at Lewis Springs and Horsethief Spring. Both filtered and unfiltered samples were collected in order to understand the differences in dissolved and total concentrations of organic compounds because some organic compounds will sorb to fine particulate organic matter.

Discrete samples (volumes of water collected at a single location at one point in time) were sealed, put on ice, and then shipped to the USGS National Water Quality Laboratory for analysis. In addition to discrete samples, in 2008 passive sampling devices (polar organic chemical integrative samplers (POCIS) and semipermeable membrane devices (SPMD)) were deployed at Murray Springs. Passive samplers can detect compounds at lower concentrations than discrete sampling (Alvarez, 2010). In 2009, POCIS and SPMDs were deployed at Murray, Horsethief, and Lewis Springs. In 2006 and 2010, only discrete samples were acquired from Murray Springs, and in both 2009 and 2010 discrete samples were also acquired from the EOP. A number of the 2008 trace organic compound (TOC) detections were censored due to contamination in the blank samples (POCIS, 9 detections, and SPMD, 16 detections; see table A1).



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: -111.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Figure 47. Map showing the location of four subwatershed springs (Moson Spring, Lewis Springs, Murray Springs, and Horsethief Spring) and a flowing well (McDowell-Craig Farm) monitored quarterly by the U.S. Geological Survey in the Sierra Vista Subwatershed, southeastern Arizona.

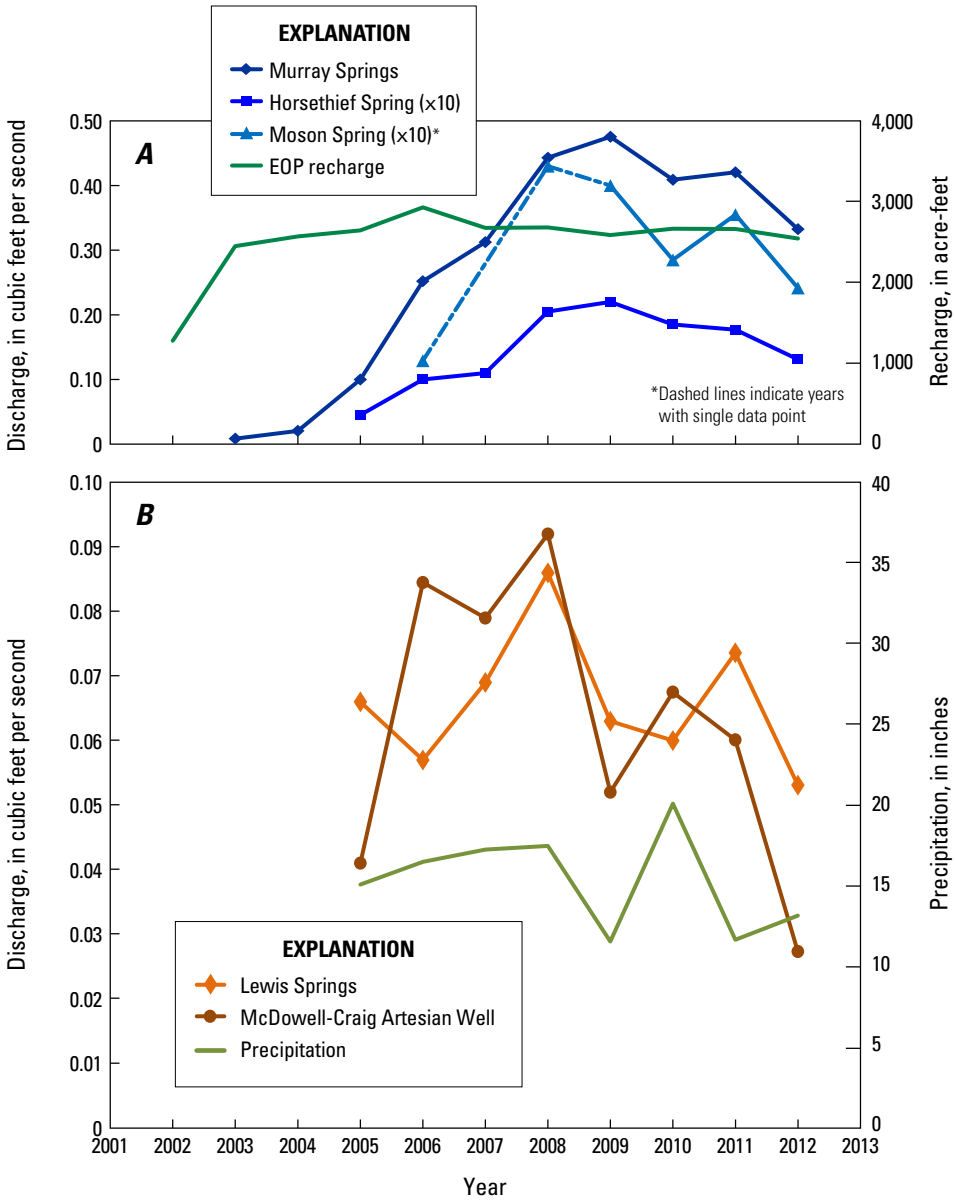


Figure 48. Graph of west-side median spring discharge and City of Sierra Vista's Environmental Operations Park (EOP) recharge along the upper San Pedro River compared to east-side median spring discharge and four-station annual precipitation in the Sierra Vista Subwatershed, southeastern Arizona. *A*, Annual median spring discharge for springs on the west side of the river (Murray Springs, Horsethief Spring, and Moson Spring) and annual EOP recharge through 2012. To enhance intercomparison of annual values, Horsethief Spring and Moson Spring discharge values have been increased by a factor of 10 from actual recorded discharge. In 2006 and 2008, Moson Spring discharge was measured once (dashed line); 2006 and 2008 Moson Spring discharge values may not be representative of entire year. *B*, Annual median spring discharge for Lewis Springs and McDowell-Craig Farm artesian well on the east side of the river and four-station annual precipitation, 2005 to 2012.

Indicator 14. Springs Water Quality—Results

Trace-organic concentrations detected by passive sampling techniques (POCIS and SPMD) are found in table A1. In 2006, only tributylphosphate was detected in the discrete sample at Murray Springs (general use of organic compounds discussed in this section are listed in table A2). The number of TOC detections at Murray Springs increased in the 2008 discrete sample, when 7 TOCs were detected in the discrete sample and 10 TOC detections were determined in one or both of the POCIS and SPMD samples. Several pharmaceutical compounds, such as phenobarbital, phenytoin, oxycodone, and codeine were detected in the passive samples. Four pharmaceutical compounds were found in both the discrete and passive samples from Murray Springs (DEET (*N*,*N*-diethyl-*m*-toluamide), tributylphosphate, phenobarbital, and phenytoin); there were no TOCs common to both the discrete and passive samples for the noncensored detections. Three times as many TOCs were collected with the POCIS than with the SPMD, suggesting the presence of fewer polar compounds (fig. 49).

In 2009, Lewis Springs (the reference site) had the lowest number of overall detections of all springs for both the discrete and passive samples. Lewis Springs did have a detection of beta-sitosterol that exceeded the Murray Springs detection. Horsethief Spring was very similar to Murray Springs with the exception of a detection of DEET in the discrete sample. Detections in 2009 at Murray Springs for both discrete and passive samples were less than in 2008, although the types of TOCs detected at Murray Springs were different and the number of TOCs were two to three times greater than from the other springs sampled (fig. 49*A*, *B*). At Murray Springs, caffeine, carbamazepine, and tributylphosphate were detected in the 2009 discrete sample at very low levels, and iminostilbene, indole, carbazole, isophorone, and beta-sitosterol

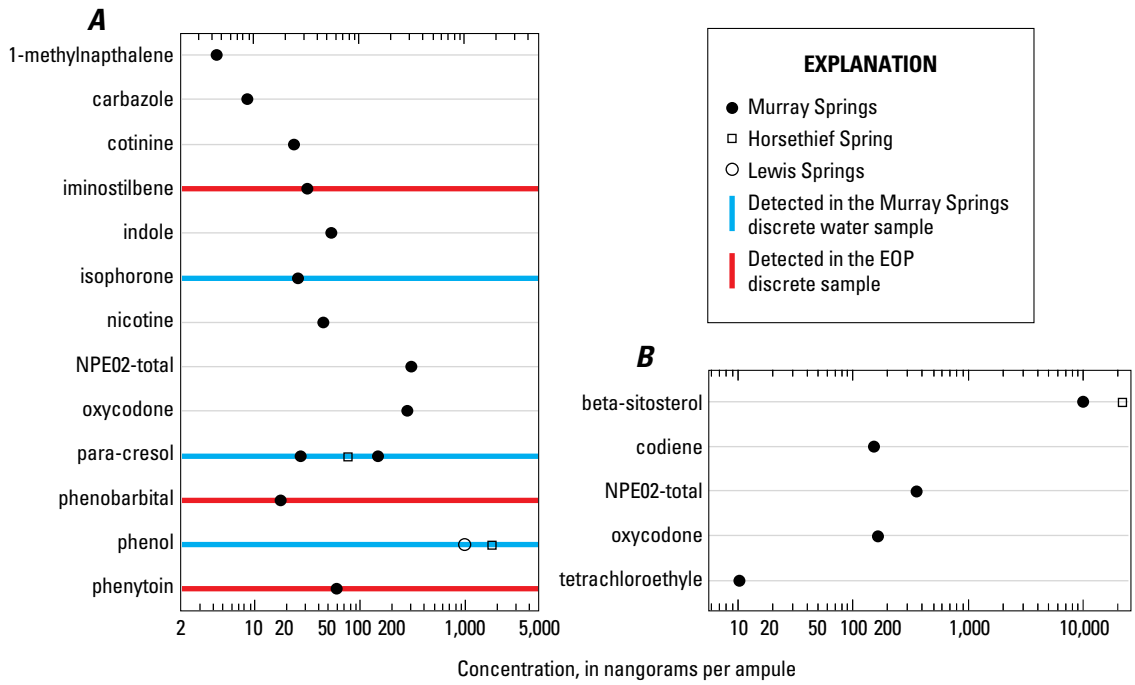


Figure 49. Plots of trace organic compound concentrations measured at sites in the Sierra Vista Subwatershed, southeastern Arizona. *A*, Polar organic chemical integrative sampler (POCIS) detections; *B*, semipermeable membrane device (SMD) detections. Samples are from Murray Springs, Horsethief Spring, and Lewis Springs. Blue lines indicate a compound detected in a Murray Springs discrete water sample; red lines indicate a compound detected in a discrete water sample at the City of Sierra Vista's Environmental Operations Park (EOP).

were detected in the passive samplers (figs. 49, 50). The EOP discrete sample contained the greatest number of TOCs (22 detections, all estimated concentrations below the laboratory reporting limit). Only three TOCs (DEET, tributylphosphate, and iminostilbene) were similar to compounds detected in the 2009 discrete and passive samples from the springs.

In 2010, only one compound (para-cresol) was detected in the discrete sample at Murray Springs. With the exception of 2-ethyl-2-phenylmalonamide, TOCs detected in the 2009 discrete dissolved sample also were detected in the 2010 total concentration sample (fig. 50). Ten more compounds were detected in the 2010 unfiltered sample than in the 2009 filtered sample, indicating that more TOCs sorb to the fine particulate organic matter than stay in the dissolved phase.

Samples collected at Murray Springs and the other nearby springs showed the presence of TOCs using both discrete and passive sampling methods (see tables A1, A3, and A4). The reference location (Lewis Springs) had the least number of detections followed by Horsethief Spring. Murray Springs, located directly downgradient of the EOP, had the greatest number of detections of all the springs, and the EOP had more than twice the detections of Murray Springs and at much higher concentrations. There were few like compounds at both the springs and the EOP, and the number of detections did not increase over the collection period as might have been expected if more water from the EOP was discharging at Murray Springs. The number of TOCs detected at Murray Springs peaked at seven discrete sample detections in 2008 and decreased in 2010 to a single detection in the discrete sample (see table A3).

As expected, overall fewer TOCs were detected in the discrete samples, whereas many more were detected with the passive samplers. The greater number of detections from passive samplers compared to discrete samples is likely related to the greater mass of analyte collected by the passive samplers over the 3-week deployment. Passive samplers provide an integrated sample over an extended period of time (several weeks to months) and range of hydrologic conditions. Passively collected samples, therefore, are more likely than discrete samples to provide a detectable concentration of episodically discharged TOCs. In addition, the integrative nature of passive samplers accumulates TOCs and increases the probability that TOC concentrations will be above method detection limits.

Limitations of the study prevent the determination of the water concentration of passive samplers and also prevent linking TOCs detected at the EOP with compounds detected from the springs. TOCs could also derive from other sources such as septic systems. Many studies have shown that TOC concentrations for POCIS samples are strongly controlled by the POCIS sampling rate for the TOC, which can be difficult to define because of many variables, including water velocity, water temperature, and biofouling (Alvarez, 2010; Alvarez and others, 2007; Morin and others, 2012). Several previous

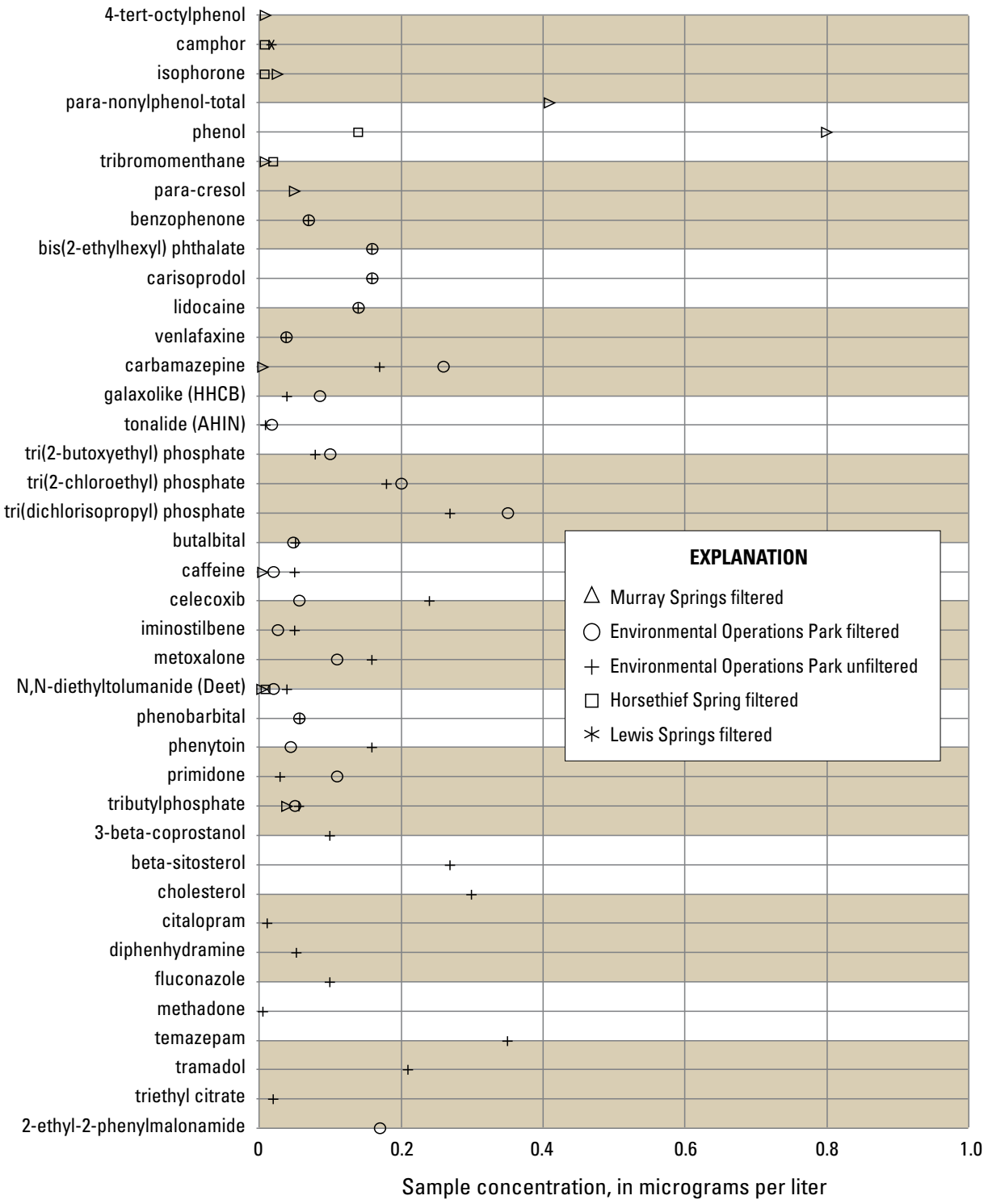


Figure 50. Plot of organic-compound concentrations in discrete samples collected at Murray Springs (2006, 2008, 2009, and 2010), Horsethief Spring (2009), Lewis Springs (2009), and the City of Sierra Vista's Environmental Operations Park (2009 and 2010) in the Sierra Vista Subwatershed, southeastern Arizona.

studies have compared TOCs in POCIS samples to discrete samples under field conditions (Alvarez and others, 2004; Alvarez and others, 2005; Zhang and others, 2008; Kolpin and others, 2013). POCIS samples generally had an equal or greater number of detections of TOCs than discrete samples.

Another consideration is the distance and basin substrate through which effluent is transported downgradient over the 0.7-mi distance from the EOP to Curry Draw. The fate of TOCs is dependent on several factors such as sorption, degradation, and transformation. The subsurface material acts like a filter where the processes mentioned can act on the TOCs, but similar to the function of a filter, the sorption mechanism can be finite, potentially making TOCs more soluble. Information about the fate and transport of these types of organic compounds in basin material is not very well understood.

Although the use of passive samplers, specifically POCIS and SPMD, to sample TOCs in water has grown in popularity in recent years (Alvarez and others, 2004; Petty and others, 2004; Alvarez, 2010; Rosen and others, 2010, Richardson and Ternes, 2011; Morin and others, 2012; Kolpin and others, 2013), the processing steps for the passive samplers require large quantities of solvent that can be susceptible to atmospheric contamination. A majority of the compounds detected had to be censored due to contamination in the blanks, and several compounds were detected in the blanks that were not detected in the environmental samples. Future passive sampling studies should emphasize quality control to identify sources and confidence intervals for censoring thresholds to ensure accurate results.

Discussion

Groundwater and surface-water resources are intricately linked. Groundwater extraction reduces the amount of groundwater that flows to riparian areas and streams, and can affect other downstream users, including humans and natural systems (Barlow and Leake, 2012). A sustainable pumping rate for a region requires local stakeholders to determine the balance between the benefits of groundwater use versus the impacts on riparian systems and streams and other downstream users. Through various proxies, including designation of the San Pedro Riparian National Conservation Area by the U.S. Congress, society has determined that permanent maintenance of this riparian system is a valid and important use for groundwater in the Sierra Vista Subwatershed. Sustainable pumping rates are thus constrained by the need to permanently maintain groundwater levels and hydraulic gradients that will, to some degree, maintain the upper San Pedro River riparian system.

Rates and locations of groundwater pumping and recharge in combination with possible climate-change effects affect different indicators differently. The magnitude of the effect depends on indicator sensitivity, the proximity of the indicator monitoring to the pumping or recharge, and the interconnectedness of the groundwater and surface-water resources. For example, water levels across much of Fort Huachuca have been in decline since at least the 1990s, whereas water levels on the east side of the San Pedro River have not. Pumping in the Sierra Vista and Fort Huachuca areas has continued without a decline in streamflow permanence at Charleston, whereas base-flow discharge at Charleston has declined. Although base flow is a more sensitive indicator of sustainability, streamflow permanence is much easier and less costly to observe. Streamflow permanence appears to be less sensitive to changes in annual streamflow than the base-flow indicator, but many more streamflow-permanence monitoring locations can be established than can streamflow gaging stations necessary for calculating base flow.

Table 5 summarizes the indicator trends observed in this report. Indicator trends were interpreted as increased, stable, or decreased groundwater discharge to the San Pedro River and adjacent riparian area is occurring or is imminent. Figure 51 shows the spatial representation of the indicator trends from table 5. Although the spring and regional-well data likely represent conditions across a broader area than the 0.5-mi diameter circles shown in figure 51, there was no simple way to accurately determine the true representative area and display it accurately. Indicators that could not contribute meaningful information regarding groundwater discharge trends to the riparian area and (or) the river are not included in figure 51 (indicators 3, 4, 11, and 14; table 5). Although indicator 4, the annual water budget balance, is characterized as a subwatershedwide indicator, the vast majority of recharge and discharge that affects that value occurs in a few locations—pumping centers and areas of natural and artificial recharge—and so there was no simple way to display this indicator's spatial representation. Its individual components are characterized by other indicators as well.

Indicator trends as shown in table 5 and figure 51 suggest declining conditions were occurring downgradient (northeast) of the pumping centers of Sierra Vista and Fort Huachuca. Regional aquifer wells across Fort Huachuca's East Range have dropped steadily since the mid-1990s, as have the horizontal gradients that drive groundwater flow to the San Pedro River (table 5 and fig. 51). Base-flow discharge at all of the subwatershed gaging stations on the San Pedro River (Palominas, Charleston, Tombstone) and the Lower Babocomari River gaging stations was declining, and stable-isotope data from near the Lower Babocomari River gaging station suggested that some capture of groundwater discharge to that stream may have been occurring. Wet-dry mapping indicated that reach 9 (Fairbank South) was almost completely dry in late June from 2004–2012, reach 10 (Fairbank North) may be getting drier, and reach 11 (Tombstone) has been dry in late June from 1999–2012 with the exception of 2001. The decline in winter precipitation beginning in about 2001 may have been in part responsible for some of these trends, although Fort Huachuca water levels and horizontal gradients had been in fairly steady decline since at least the mid-1990s.

However, not all indicators in the region downgradient of the pumping centers suggested decreasing discharge to the riparian area and the San Pedro River (table 5 and fig. 51). Near-stream vertical gradients in the Moson, Boquillas, and Fairbank

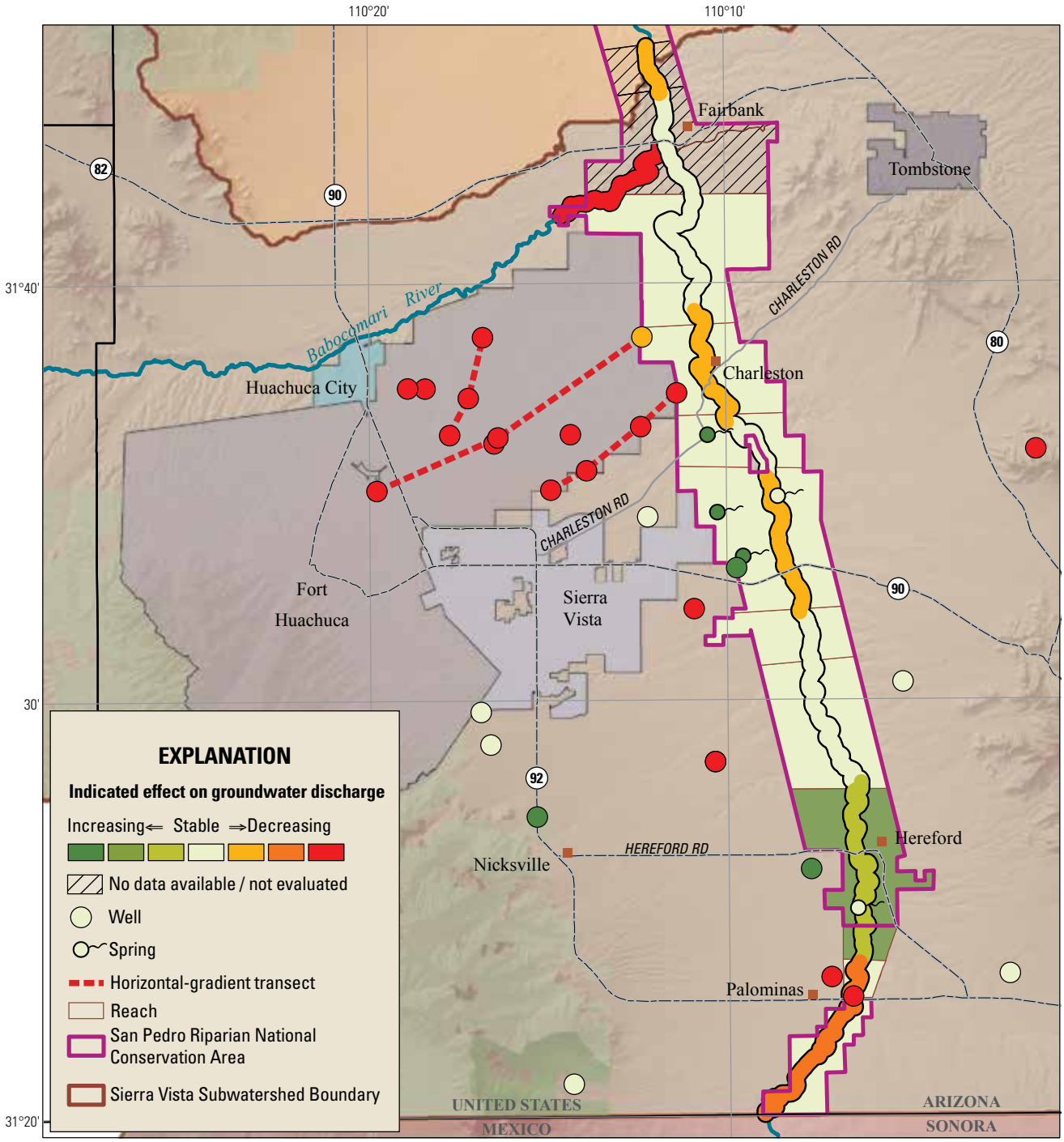
Table 5. Evaluation of indicator trends for groundwater discharge to the riparian area and (or) river in the Sierra Vista Subwatershed, southeastern Arizona.

[Water-quality data are unrelated to groundwater discharge trends. Period of record is typically 10 years or more. R, reach. These results are shown spatially in figure 51]

Indicators of sustainable groundwater use for the Sierra Vista Subwatershed			
Increasing discharge	**Stable discharge**	**Decreasing discharge**	**Insufficient data**
1. Regional aquifer water levels			
LSP-6	Antelope Run #2	Moncrief #1	
Holder	Antelope Run #3	Palominas (deep)	
Rambo	Baumkirchner 3	Palominos Fire	
	Bella Vista	Ranch Road Deep	
	Foudy	SB Ranch	
	Misty Ray Road/Granite	TW1	
	MW7	TW4	
		TW6	
		TW7	
		TW8	
		TW9	
		TW9W	
		MW1	
		MW2	
		MW3	
		MW4	
		MW5	
		MW6	
2. Horizontal hydraulic gradients (regional)			
		Fort Huachuca northwest	
		Fort Huachuca central	
		Fort Huachuca southeast	
3. Aquifer-storage change—microgravity			
			West subwatershed
4. Annual groundwater-budget balance			
		Entire subwatershed	
5. Near-stream alluvial-aquifer water levels			
R2-Hereford	R1-Palominas		R10-Fairbank north
	R3-Hunter		
	R4-Cottonwood		
	R5-Lewis Springs		
	R7-Charleston		
6. Near-stream vertical gradients			
	R1-Palominas		
	R2-Hereford		
	R4-Cottonwood		
	R5-Lewis Springs		
	R6-Moson Spring		
	R8-Boquillas		
7. Annual fluctuation of near-stream alluvial-aquifer water levels			
	R1-Palominas		
	R3-Hunter		
	R4-Cottonwood		
	R5-Lewis Springs		
	R7-Charleston		

Table 5.—Continued

Indicators of sustainable groundwater use for the Sierra Vista Subwatershed			
Increasing discharge	**Stable discharge**	**Decreasing discharge**	**Insufficient data**
8. Streamflow permanence			
	R1-Palominas R2-Hereford R3-Hunter R5-Lewis Springs R6-Moson Spring R7-Charleston R10-Fairbank north R11-Tombstone		R8-Charleston Mesquite
9. Base flow in San Pedro and Babocomari Rivers			
		R1-Palominas R7-Charleston R8-Lower Babocomari R11-Tombstone	
10. Wet-dry status			
R2-Hereford	R1-Palominas R3-Hunter R4-Cottonwood R5-Lewis Springs R6-Moson R7-Charleston R8-Charleston Mesquite/ Boquillas R9-Fairbank south R10-Fairbank north R11-Tombstone		
11. San Pedro River water quality			
			Charleston
12. San Pedro and Babocomari Rivers isotope analysis			
	R2-Hereford R7-Charleston R11-Tombstone	R1-Palominas R5-Lewis Springs R8-Lower Babocomari	
13. Springs discharge			
Horsethief Spring Murray Springs Moson Spring	McDowell-Craig well Lewis Springs		
14. Springs water quality			
			Horsethief Spring Murray Springs Lewis Springs



Base map from U.S. Geological Survey digital data, various scales; roads from U.S. Census Bureau with Arizona State Land Department; boundaries from Arizona State Land Department, Bureau of Land Management, Salt River Project, and U.S. Geological Survey.

0 3 6 MILES
0 3 6 9 12 KILOMETERS

Coordinate System: NAD 1983 UTM Zone 12N
Projection: Transverse Mercator
Datum: North American 1983
False Easting: 500,000.0000
False Northing: 0.0000
Central Meridian: -111.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Figure 51. Map showing spatial representation of Sierra Vista Subwatershed, southeastern Arizona, hydrologic trends in 2012 based on 11 of the 14 indicators of sustainable groundwater use shown in table 5 (indicators 3, 4, 11, and 14 are not included). Dark green represents increasing groundwater discharge to the riparian area and (or) San Pedro River; pale green represents stable groundwater discharge to the riparian area and (or) river; red represents declining groundwater discharge to the riparian area and (or) river. Lighter shades of green or red represent river or riparian reaches where more than one indicator has been evaluated with differing results; the representation is an average of the combined indicator trends.

North reaches were stable overall, alluvial aquifer water levels and water-level fluctuations in the Charleston reach, streamflow permanence in the Charleston and Tombstone reaches, wet-dry lengths in the Charleston, Boquillas-Charleston Mesquite, and Fairbank North reaches, and isotope data from the Charleston and Tombstone reaches all remained stable. Nevertheless, it is a hydrologic certainty that without significant mitigation, impacts on the San Pedro River and San Pedro riparian area from Charleston to Tombstone will eventually become evident (Leake and others, 2005; Barlow and Leake, 2012). Water quality at Charleston showed few trends (although suspended sediment has greatly decreased since the 1970s with a late exception).

In other parts of the subwatershed, indicators were either mixed or stable (table 5 and fig. 51). For example, water levels appeared to be in decline in the unregulated subdivisions and private wells south-southeast of the City of Sierra Vista and away from the river, whereas regional-aquifer wells closer to the river had recovered. Vertical gradients, water levels, and water-level fluctuations in the alluvial aquifer near the river in that same general area (Hunter and Cottonwood, reaches 3 and 4) were stable. Base-flow discharge at Palominas gaging station (reach 1) appeared to be in decline, and increasing trends in stable isotopes suggested a declining groundwater contribution in that area may be a contributor. Other indicators for this reach were stable (alluvial-aquifer water levels and fluctuations, vertical gradients, streamflow permanence, wet-dry status). Immediately downstream, all indicators evaluated for the Hereford reach had remained stable over their period of record, with the exception of a statistically significant increase in wetted length in the June dry season from 1999 to 2012 and a statistically significant increase in alluvial aquifer dry-season water levels when comparing the second half of the period of record (2007–12) to the first half of the period of record (2001–06), suggesting there had been an improvement in conditions there.

The expanding cone of depression (as expressed by the declining horizontal hydraulic gradients and decreasing water levels on Fort Huachuca) should be of interest to water managers and to those with an interest in the SPRNCA. Even if pumping were immediately reduced or stopped, the cone would continue to propagate for decades or more (Leake and others, 2005; Barlow and Leake, 2012). Without significant mitigation measures, it is likely too late already to prevent declining water levels from reaching the San Pedro River riparian area from Charleston to Tombstone. Ongoing evaluation of the indicators in this report can help determine the rate at which effects are propagating toward the river and are increasing within the riparian area, and when and where to develop and implement interim measures to mitigate deteriorating hydrological conditions. In addition, monitoring data from the indicator sites is necessary to refine and improve the regional groundwater model from which better, more accurate projections can be made regarding the sustainability of various rates and locations of groundwater withdrawal, as well as regarding the mitigating effects of various rates and locations of enhanced recharge in the subwatershed. Capture mapping done by Leake and others (2008; see also Barlow and Leake, 2012) provides a good first cut for such evaluations, and subsequent modeling efforts can then provide greater specificity (for example, Lacher and others, 2014).

The Upper San Pedro Partnership, as well as a few member groups acting individually, has water-conservation and water-management projects currently under way and others in the planning stages. For example, an ongoing water-conservation and education campaign has likely been at least partially responsible for reducing the overall per capita rate of groundwater use in the subwatershed (Upper San Pedro Partnership, 2015). This helps to reduce the effects of groundwater pumping that are already propagating through the system. Cochise County has recently completed a stormwater recharge project just north of Palominas, which turns local surface runoff that mostly left the subwatershed as San Pedro River storm flow into groundwater recharge that can then more slowly discharge to the riparian area over months (Lacher and others, 2014; Richter and others, 2014). The Nature Conservancy is testing the feasibility of similar projects further downstream (Brooke Bushman, The Nature Conservancy, written commun., 2015). Such short-term solutions are intended to directly supply the riparian system with water and may partially compensate for the reduced groundwater discharge to the riparian area as the cone of depression expands through the northernmost reaches of the subwatershed (Charleston to Tombstone).

The potential volume of such recharge projects depends on the size of the watershed(s) upstream from a given project site, proximity to urbanized (impervious) areas, and the recharge-site characteristics. Potential recharge may range from a few percent to more than 20 percent of annual precipitation over the recharge site (Mike Milczarek, GeoSystems Analysis, Inc., written commun., 2015). On average, stormwater recharge projects in the subwatershed have averaged less than 100 acre-ft of recharge annually. Modeling of hypothetical recharge sites near Palominas, the mouth of Garden Canyon, and along the lower Babocomari River suggests that about 2,400 acre-ft of near-stream recharge distributed annually across these three hypothetical locations could maintain San Pedro River base flows at 2003 levels for as many as 100 years given expected population trends (Lacher, 2012; Lacher and others, 2014). Where such water might come from is unknown at this time.

Future Work

Every indication is that a comprehensive monitoring program will continue across the Sierra Vista Subwatershed well into the future. Follow-up reports that build on the framework developed in this initial, comprehensive assessment of sustainability should thus be possible and relatively simple to produce, because much of the background and methods presented here would not need to be reproduced annually.

An alternative to annual reporting might be a Web-based data and information portal where plots and other indicator-based information-graphics could be updated automatically as new data are collected and published. Dynamic, easily understandable, and readily available information-based graphics would ensure that the data gathered across the subwatershed are of interest and value to those most interested in regional water issues and the San Pedro

River riparian area. At the same time, the data on which these information-based graphics are based would be readily available to researchers and the public alike and within a reasonable time following their collection.

Water-budget data are the most difficult, wide-ranging, and labor-intensive indicator data to collect. It may be worthwhile to evaluate the subwatershed groundwater budget balance on a less frequent basis than other indicators, as it is less precise and thus less likely to vary beyond its estimated uncertainty from one year to the next. It may also be worthwhile to replace some of the near-stream alluvial-aquifer wells that are used for water levels, vertical gradients, and water-level fluctuations with other, existing SPRNCA wells. The advantage would be to obtain indicator coverage in more SPRNCA reaches. The disadvantage would be that the existing record for any added wells could be missing as many as 10 years of data, and so it would take time before trends from these well data might become evident. Such adjustments to the monitoring-and-reporting process will be important for stakeholders to discuss in the future.

Summary and Conclusions

This report represents a significant effort to summarize, analyze, and interpret a large volume of data related to the hydrological condition of the San Pedro River of southeastern Arizona and the aquifer of the Sierra Vista Subwatershed. The sustainability indicators evaluated in this report provide a snapshot of the state of groundwater conditions in the subwatershed as of 2012 and the effects of human factors (including groundwater-pumping and water-management strategies), and natural factors (including climate variability) on surface water and groundwater. Some indicators, such as regional groundwater levels, horizontal groundwater gradients, direct measurement of aquifer-storage change, and the groundwater budget, reflected subwatershedwide conditions. Others indicators evaluated the condition of the riparian system (alluvial-aquifer water levels, near-stream vertical gradients, and annual water fluctuations) using factors known to be important to the long-term health of riparian systems. The largest set of indicators (streamflow permanence, base-flow, wet-dry mapping, river water quality, and isotope analysis) assessed the state of the San Pedro and lower Babocomari Rivers themselves. The last set of indicators assessed the condition of subwatershed springs (discharge and water quality), which are significant for the aquatic and riparian ecosystems they support and are important indicators of trends in groundwater.

Many, but not all, of the indicators suggested that in 2012, hydrologic conditions in many parts of the subwatershed were stable or improving. Locations adjacent to active-management projects implemented to maintain or improve conditions, such as (1) the City of Sierra Vista's EOP treated-effluent recharge and (2) the retirement of agricultural pumping near Palominas, appeared to be responding positively with increased water levels, increased wetted river length during the dry season, and reduced annual water-level fluctuation, pointing toward the potential for continued improvements as new mitigation projects are developed and initiated. However, there are a number of indicators that reflect degrading conditions in other parts of the subwatershed. The vast majority of these sites were found downgradient from the pumping centers of Sierra Vista and Fort Huachuca and can generally be seen as expressions of the cone of depression and capture of water that would otherwise have discharged to the riparian area and near-stream alluvial aquifer near the San Pedro River. These included consistent decreases in water levels in many regional-aquifer wells near to and downgradient of the Sierra Vista and Fort Huachuca pumping centers, horizontal-gradient declines across Fort Huachuca, long-term declines in base-flow discharge at stream-gaging stations (Charleston, Tombstone, Lower Babocomari, and Palominas), increasing trends in stable isotopes indicative of decreased groundwater discharge to the Babocomari River near the Lower Babocomari gaging station and to the San Pedro River near Palominas, and two river reaches that have remained essentially dry during the driest time of the year throughout the period of record (Fairbank South and Tombstone).

Information found here can be used to help determine the kinds and locations of groundwater-management efforts needed in the subwatershed. Stabilizing groundwater discharge to the riparian area and base-flow levels in the San Pedro River will be challenging. If pumping across the subwatershed were to completely stop tomorrow, the cone of depression would continue to spread and its effects, including decreases in natural discharge to the riparian area and river, would continue for decades, even as the cone of depression filled (Leake and others, 2005; Barlow and Leake, 2012). In fact, to stabilize groundwater discharge to the riparian area and base flow in the river in spite of a spreading cone of depression would represent a major groundwater-management success.

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Appendix

Appendix—Trace Organic-Compound Concentrations and Well Names and Locations

[Tables A1 through A4 are available online only as an Excel (.xlsx) files at https://doi.org/10.3133/sir20165114]

Table A1. Trace organic-compound concentrations detected in passive samplers deployed at Murray and Lewis Springs and Horsethief Spring in the Sierra Vista Subwatershed, southeastern Arizona.

Table A2. Trace organic-compound concentrations detected in the passive sampler laboratory blanks.

Table A3. Trace organic-compound concentrations detected in discrete samples collected at Murray Springs, Lewis Springs, Horsethief Spring, and the City of Sierra Vista's Environmental Operations Plant between 2006 and 2010.

Table A4. Trace organic-compounds (wastewater compounds and pharmaceuticals) analyzed at Murray Springs, Lewis Springs, Horsethief Spring, and the City of Sierra Vista's Environmental Operations Park (EOP) in the Sierra Vista Subwatershed, southeastern Arizona.

Table A5. Monitoring-well numbers, names, and locations in the Sierra Vista Subwatershed, Arizona.

[NA, not applicable]

Site number	Site name	Common name	Location
San Pedro Riparian National Conservation Area monitoring wells			
314144110104702	D-20-21 14BCC2	BOQSUD	Boquillas
314144110104701	D-20-21 14BCC1	BOQSUS	Boquillas
314124110105801	D-20-21 15DDB	BOQ-UP	Boquillas
313738110102901	D-21-21 02DDC	CHB-LI	Charleston Bridge
313508110085702	D-21-22 19CDB2	CLRSUD	Clanton Ranch
313508110085701	D-21-22 19CDB1	CLRSUS	Clanton Ranch
313105110074501	D-22-22 17ACC3	COTBLM	Cottonwood
313110110074402	D-22-22 17ACC2	COT-LD	Cottonwood
313110110074401	D-22-22 17ACC1	COT-LS	Cottonwood
313108110075202	D-22-22 17BDD2	COTUWD	Cottonwood
314317110113401	D-21-20 03CBA	FBK-LI	Fairbank
314318110113101	D-20-21 03CAB	FBK-LO	Fairbank
314248110105901	D-20-21 10AAC2	FBK-UP	Fairbank
312716110063202	D-23-22 04DCD2	HER-LD	Hereford
312716110063201	D-23-22 04DCD1	HER-LS	Hereford
312642110063702	D-23-22 09ACD2	HERSUD	Hereford
312642110063701	D-23-22 09ACD1	HERSUS	Hereford
312716110063302	D-23-22 04DCD4	HERUED	Hereford
312716110063301	D-23-22 04DCD3	HERUES	Hereford
312859110070601	D-22-22 28CAA1	HUN-LI	Hunter
312900110071001	D-22-22 28CAA2	HUN-LO	Hunter
312853110070401	D-22-22 28CCD	HUN-SO	Hunter
312843110070801	D-22-22 33BBC	HUN-UP	Hunter
312453110061401	D-23-22 22CBB1	KOL-LI	Kolbe
312453110061301	D-23-22 22CBB2	KOL-LO	Kolbe

Table A5.—Continued

Site number	Site name	Common name	Location
312453110061101	D-23-22 22CBB3	KOL-UP	Kolbe
313312110082302	D-22-22 06AAA6	LSP-1	Lewis Springs
313312110082303	D-22-22 06AAA5	LSP-5	Lewis Springs
313310110081701	D-22-22 06AAA1	LSPLED	Lewis Springs
313312110081302	D-22-22 05BBD2	LSPUED	Lewis Springs
313643110100102	D-21-21 13BAB2	MOSLND	Moson
313643110100101	D-21-21 13BAB1	MOSLNS	Moson
313642110100102	D-21-21 13BAB4	MOSLNW	Moson
313640110100001	D-21-21 13BAC2	MOS-UD	Moson
313638110095901	D-21-21 13BAC1	MOSWAS	Moson
312208110070502	D-24-22 04CBA2	PAL-LD	Palominas
312226110062101	D-24-22 04ADD	PAL-SA	Palominas
312207110070702	D-24-22 04CBA4	PAL-UD	Palominas
312214110071602	D-24-22 04CBC2	PALUWD	Palominas
312214110071601	D-24-22 04CBC1	PALUWS	Palominas
312127110073101	D-24-22 08DBA1	PLS-LI	Palominas South
312127110073301	D-24-22 08DBA2	PLS-LO	Palominas South
314511110120601	D-19-21 28ACD	TOM-LI	Tombstone
Fort Huachuca regional monitoring wells			
313721110112401	D-21-21 10ADA1	MW-1	Fort Huachuca
313729110185601	D-21-20 09ABC1	MW-2	Fort Huachuca, East Range
313502110145701	D-21-21 19CDD1	MW-3	Fort Huachuca, East Range
313530110135601	D-21-21 20ACC1	MW-4	Fort Huachuca, East Range
313633110122501	D-21-21 16AAD1	MW-5	Fort Huachuca, East Range
313610110163201	D-21-20 14DAD1	MW-6	Fort Huachuca, East Range
313841110122201	D-20-21 33CDD1	MW-7	Fort Huachuca, East Range
313507110195101	D-21-20 20DCD1	TW-1	Fort Huachuca, East Range
313615110175601	D-21-20 15ACD1	TW-4	Fort Huachuca, East Range
313845110165201	D-20-20 35CCB1	TW-6	Fort Huachuca, East Range
313706110171601	D-21-20 11BCD1	TW-7	Fort Huachuca, East Range
313618110162301	D-21-20 13CBD1	TW-8	Fort Huachuca, East Range
313622110142101	D-21-21 17BCC1	TW-9	Fort Huachuca, East Range
313729110182501	D-21-20 10BBD1	TW-9W	Fort Huachuca, East Range
Other Sierra Vista Subwatershed regional monitoring wells			
312856110163801	D-22-20 26DDC	Antelope 2	South of Sierra Vista
312943110165401	D-22-20 26ABB1	Antelope 3	South of Sierra Vista
312044110141901	D-24-21 17BCB3	Baumkirchner 3	Coronado National Memorial
313421110121401	D-21-21 27CBD	Bella Vista	East of Sierra Vista
312323110020901	D-23-23 32BCB	Foudy	East of San Pedro River
312555110074301	D-23-22 17ACC	Holder	Southeast of Sierra Vista
313309110094301	D-22-22 06ABD	LSP-6	Lewis Springs
313024110050701	D-22-22 23BCA	Misty Ray	East of San Pedro River
313558110012001	D-21-23 17DDA	Moncreif #1	East of San Pedro River
312250110063901	D-23-22 33DCD2	PAL Deep	Route 92 and San Pedro River

Table A5.—Continued

Site number	Site name	Common name	Location
312318110071901	D-23-22 33BCC	Palo Firehouse	North of Palominas
312712110152101	D-23-21 06CCC2	Rambo	South of Sierra Vista
312830110102301	D-22-21 35ADD1 PZ1	Ranch Deep	Southeast of Sierra Vista
312830110102302	D-22-21 35ADD1 PZ2	Ranch Shallow	Southeast of Sierra Vista
313210110105701	D-22-21 11BDA1	SB Ranch	East of Sierra Vista

Menlo Park Publishing Service Center
Manuscript approved for publication August 2, 2016
Edited by James W. Hendley II
Layout by Cory D. Hurd

ISSN 2328-0328 (online)
https://doi.org/10.3133/sir20165114