In cooperation with the City of Austin

Concentrations, Loads, and Yields of Particle-Associated Contaminants in Urban Creeks, Austin, Texas, 1999–2004

Scientific Investigations Report 2006–5262

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

Cover.
Front: Stormflow at U.S. Geological Survey streamflow-gaging station Barton Creek at Lost Creek Boulevard near Austin (photograph by Toby Welborn, U.S. Geological Survey).

Back Top: Stormflow at U.S. Geological Survey streamflow-gaging station Barton Creek above Barton Springs at Austin (photograph by Marcus Gary, U.S. Geological Survey).

Back Bottom: Stormflow at Barton Creek at Lost Creek Boulevard, Austin (photograph by Toby Welborn, U.S. Geological Survey).

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U.S. Geological Survey
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Conversion Factors

Inch/Pound to SI

Multiply	By	To obtain
	Length	
inch (in.)	25.4	millimeter (mm)
mile (mi)	1.609	kilometer (km)
yard (yd)	0.9144	meter (m)
	Area	
acre	0.4047	hectare (ha)
square mile (mi^2)	2.590	square kilometer (km^2)
	Flow rate	
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
inch per hour (in/hr)	25.4	millimeter per hour (mm/hr)
inch per second (in/s)	25.4	millimeter per second (mm/s)

SI to Inch/Pound

Multiply	By	To obtain
	Length	
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
micrometer (µm)	3.937×10^{-5}	inch (in.)
millimeter (mm)	0.03937	inch (in.)
	Mass	
gram (g)	0.03527	ounce avoidupois (oz advp)
microgram (µg)	3.527×10^{-8}	ounce avoidupois (oz advp)
milligram (mg)	3.527×10^{-5}	ounce avoidupois (oz advp)
	Volume	
liter (L)	0.2642	gallon (gal)
milliliter (mL)	0.034	fluid ounce (fl oz)

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

$$°F = 1.8\ °C + 32$$

Abbreviations:

microgram per liter, µg/L
microgram per kilogram, µg/kg
milligram per kilogram, mg/kg
milligram per liter, mg/L

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By B.J. Mahler, P.C. Van Metre, J.T. Wilson, A.L. Guilfoyle, and M.W. Sunvison

Abstract

Concentrations, loads, and yields of particle-associated (hydrophobic) contaminants (PACs) in urban runoff in creeks in Austin, Texas, were characterized using an innovative approach: large-volume suspended-sediment sampling. This approach isolates suspended sediment from the water column in quantities sufficient for direct chemical analysis of PACs. During 1999–2004, samples were collected after selected rain events from each of five stream sites and Barton Springs for a study by the U.S. Geological Survey, in cooperation with the City of Austin. Sediment isolated from composited samples was analyzed for major elements, metals, organochlorine compounds, and polycyclic aromatic hydrocarbons (PAHs). In addition, at the Shoal Creek and Boggy Creek sites, individual samples for some events were analyzed to investigate within-event variation in sediment chemistry. Organochlorine compounds detected in suspended sediment included chlordane, dieldrin, DDD, DDE, DDT, and polychlorinated biphenyls (PCBs). Concentrations of PACs varied widely both within and between sites, with higher concentrations at the more urban sites and multiple nondetections at the least-urban sites. Within-site variation for metals and PAHs was smaller than between-site variation, and concentrations and yields of these and the organochlorine compounds correlated positively to the percentage of urban land use in the watershed. Loads of most PACs tested correlated significantly with suspended-sediment loads. Concentrations of most PACs correlated strongly with three measures of urban land use. Variation in suspended-sediment chemistry during runoff events was investigated at the Shoal and Boggy Creek sites. Five of the eight metals analyzed, dieldrin, chlordane, PCBs, and PAHs were detected at the highest concentrations in the first sample collected at the Shoal Creek site, a first-flush effect, but not at the Boggy Creek site. Temporal patterns in concentrations of DDT and its breakdown products varied from one event to the next. In spite of the first-flush effect in concentrations at the Shoal Creek site, most of the contaminant load was transported at peak discharge, when suspended-sediment concentration and load are maximum.

Introduction

From 1990 to 2000, Austin, Tex., was the fifth fastest-growing city in the Nation, increasing in population by 48 percent (Scheibal and Glass, 2001). The conversion of nonurban to urban land uses usually causes water quality to deteriorate (Rice, 1999; Van Metre and others, 2000; Wakeham and others, 1980). Rapid growth has raised concerns among water-resource managers about the effect that urbanization is having on water quality in Austin. Chlordane, DDT, and polychlorinated biphenyls (PCBs) have been detected in Town Lake[1] bottom sediment at levels of concern for aquatic life (Van Metre and Mahler, 1999) and, historically, in fish at levels of concern for consumption (Texas Department of Health, 1999); polycyclic aromatic hydrocarbons (PAHs) in a sediment core from Town Lake increased about five-fold from the 1960s to the late 1990s (Van Metre and Mahler, 1999). Particle-associated (hydrophobic) contaminants (PACs) such as these present a threat to biota and to humans because many have one or more of the characteristics of persistence, bioaccumulation, or toxicity (U.S. Environmental Protection Agency, 1997).

The importance of suspended sediment in understanding the sources, transport, and fate of many chemical constituents in streams has long been recognized. In the summary proceedings of a 1982 U.S. Geological Survey (USGS) Sediment Chemistry Workshop, Bradford and Horowitz (1988, p. 1) stated, “The strong association of numerous toxic chemicals—both organic (such as PCBs, DDT, Mirex, and Kepone) and inorganic (such as arsenic, cadmium, lead, and mercury)—with sediment means that much of the downstream transport of these materials cannot be detected or evaluated solely through the sampling and

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[1] Colorado River in central Austin is known as Town Lake.

analysis of water." In summarizing research from a long-term USGS study of the Mississippi River, Garbarino and others (1995, p. 55) stated, "Like most of the heavy metals in the Mississippi River, lead is transported largely in association with the suspended sediment." In the same report, Rostad and others (1995, p. 103) concluded, "Many of the sparingly soluble organic contaminants found in the Mississippi River, such as hexachlorobenzene and PCBs, are associated with the suspended sediment."

PACs constitute a major water-quality issue. The Federal Agency for Toxic Substances and Disease Registry and the U.S. Environmental Protection Agency in 2005 published a Priority List of Hazardous Substances as a requirement of the Superfund program (Agency for Toxic Substances and Disease Registry, 2005). Fifteen of the top 20 contaminants on the 2005 list are hydrophobic, including four metals and 11 organic contaminants. Among these contaminants are the four responsible for the great majority of fishing advisories and fish-consumption bans in the United States—mercury, chlordane, DDT, and PCBs (U.S. Environmental Protection Agency, 2002). Also included on the list are PAHs, a group of carcinogenic contaminants reported to be increasing rapidly in U.S. urban lakes, including Town Lake, coincident with urban growth in watersheds (Van Metre and Mahler, 2005; Van Metre and others, 2000).

Although the importance of suspended sediment in the occurrence and transport of PACs is widely recognized, suspended-sediment chemistry rarely is analyzed routinely. The usual approach for measuring hydrophobic contaminants in urban streams is to collect and analyze whole-water (unfiltered) samples. Because whole-water samples by definition include both the water and the particles suspended in it, their chemical analysis includes constituents that are attached to suspended particles as well as dissolved in the water. If the suspended-sediment concentration (SSC) is less than several hundred milligrams per liter, this approach often results in nondetection of PACs, particularly organochlorine compounds and PAHs. This can occur even in cases where the concentration of contaminants associated with the suspended sediment exceeds the probable effect concentration (PEC) (Mahler and Van Metre, 2003), the consensus-based concentration above which adverse biological effects are expected (MacDonald and others, 2000). Thus, while whole-water sampling remains the approach of choice for dissolved constituents such as nutrients and polar (soluble) pesticides, an alternative approach is desirable for hydrophobic contaminants.

An alternative to whole-water sampling is to isolate and analyze suspended sediment directly through large-volume suspended sediment (LVSS) sampling. Two general approaches have been used to do this: centrifugation (Horowitz, 1995) and filtration (Mahler and Van Metre, 2003). In either case, the sediment isolated is analyzed using laboratory methods typically used for bottom sediment and the results reported as mass per unit mass dry weight (for example, micrograms per kilogram or parts per billion). This approach can be more effective for investigation of hydrophobic contaminants than analysis of whole water (which contains both water and sediment) because concentrations of hydrophobic contaminants per mass are orders of magnitude greater in the sediment than in the unfiltered water. As a result, nondetections are rare, which provides a better understanding of occurrence. To investigate the occurrence and transport of PACs in Austin urban creeks, the USGS, in cooperation with the City of Austin, conducted a study using the sampling and analysis of suspended sediment to characterize the occurrence and to estimate loads and yields of numerous PACs in urban stormwater from selected creeks in Austin.

Although analysis of streambed sediment is another approach for investigating hydrophobic contaminants, analysis of suspended sediment fulfills different objectives. Concentrations of contaminants in streambed sediment reflect the conditions to which bottom-dwelling biota (for example, benthic invertebrates and bottom fish) at a particular site and time are exposed, whereas concentrations of contaminants in suspended sediment give a more integrated view of water quality in the stream as a whole and its relation to land use in the watershed. Further, results of analyses of suspended sediment, in conjunction with flow data, can be used to compute contaminant loads and yields, whereas analyses of streambed sediment cannot.

Purpose and Scope

The primary purpose of this report is to

1. Report concentrations and associated computed loads and yields of PACs, specifically metals, organochlorine compounds, and PAHs, in transport in the urban creeks studied after rain events (storms);
2. Present relations between concentrations and loads of PACs and different explanatory variables;
3. Interpret the results in the context of existing local and nationwide data; and
4. Evaluate LVSS sampling in relation to other approaches.

This report presents the results of a study to characterize the occurrence of PACs in four urban creeks—Barton Creek, Shoal Creek, Boggy Creek, and Williamson Creek—and Barton Springs (fig. 1). This study is the first to collect and analyze suspended sediment from Austin area creeks and Barton Springs for a broad range of hydrophobic contaminants. It did not include analysis of bed sediment or sediment moving down creeks as bed load.

Climatic conditions such as rainfall depth and intensity and antecedent conditions could affect concentrations and loads of PACs. Statistical analyses were used to investigate relations between PACs (concentrations and loads) and total rainfall, average rainfall intensity, maximum rainfall intensity, antecedent dryness conditions, and antecedent flow associated with each rain event. The statistical analyses used were correlation, in cases where there were no nondetections of a PAC at a site, and survival analysis, in cases where a PAC was not detected in one or more samples at a site.

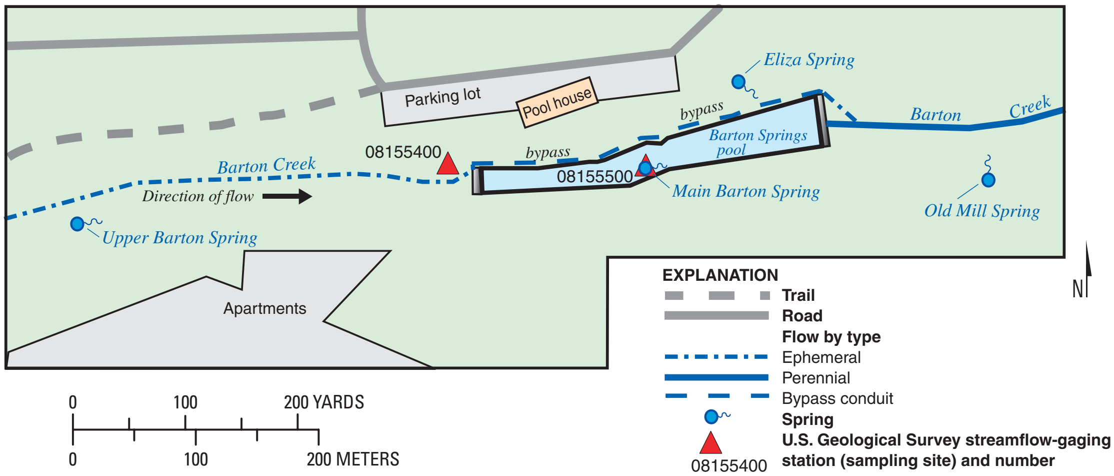


Diagram of the immediate Barton Springs area.

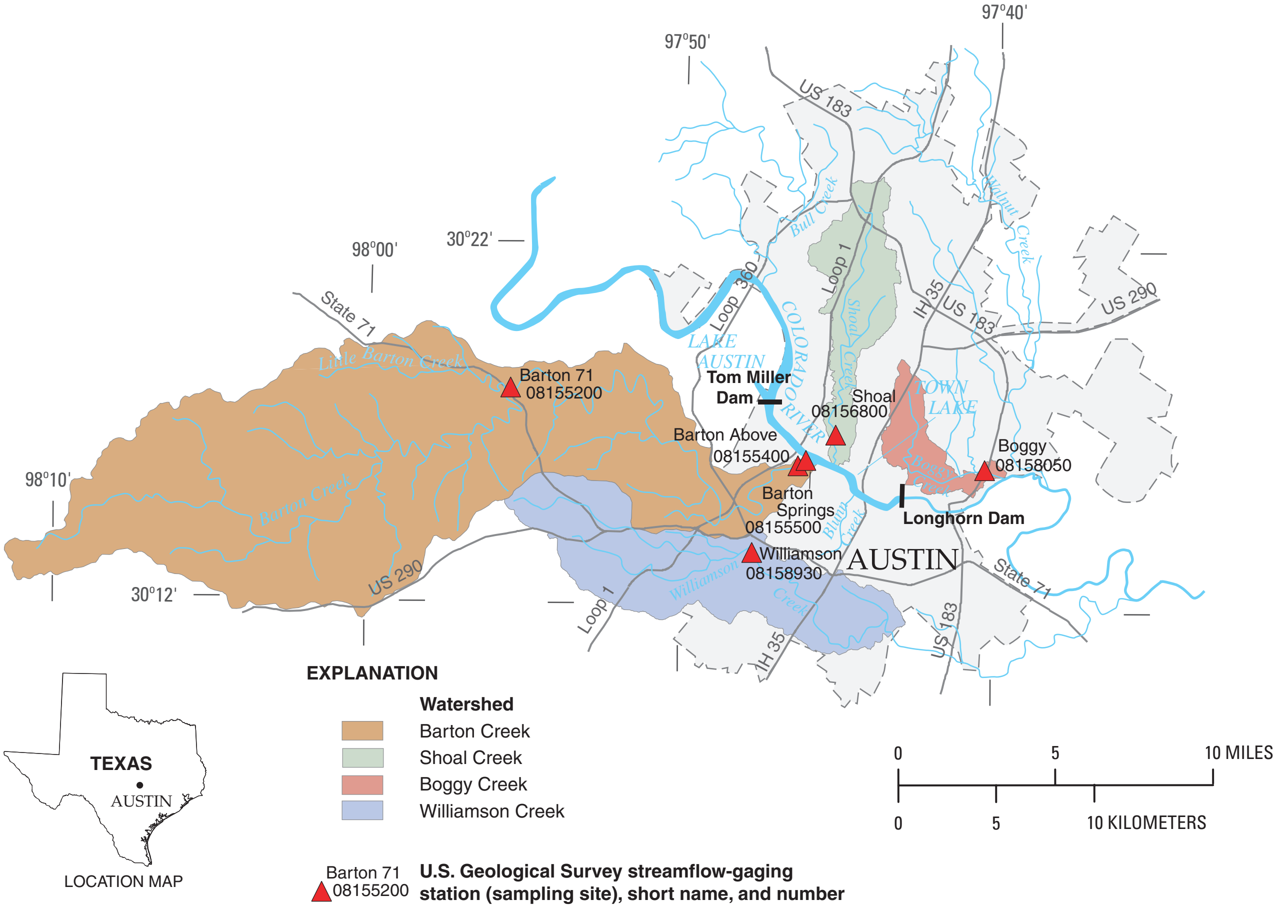


Figure 1. Sampling sites and associated watersheds, Austin, Texas.

Table 1. Sampling sites and characteristics of watersheds of creeks sampled, Austin, Texas, 1999–2004.

[USGS, U.S. Geological Survey; SF, single family; MF, multifamily; AG, agriculture; COMM, commercial; OFF, office; IND, industrial; CIV, civic; PRK, park; TRANS, transportation; UND, undeveloped; SH, State Highway; --, not applicable; US, U.S. Highway]

USGS streamflow-gaging station (number)	Sampling site short name (fig. 1)	Water-shed area (acres)	Distance to Town Lake (miles)	Land-use type (percentage of watershed)									
				SF	MF	AG	COMM	OFF	IND	CIV	PRK	TRANS	UND
Barton Creek at SH 71 near Oak Hill (08155200)	Barton 71	57,080	33.7	22.9	1.9	25.3	0.3	0.4	0.3	0.1	18.7	2.5	27.5
Barton Creek above Barton Springs at Austin (08155400)	Barton Above	76,100	1.15	22.3	2.2	21.9	.7	.9	.4	.5	22.8	4.0	24.3
Barton Springs at Austin (08155500)	Barton Springs	--	1.08	--	--	--	--	--	--	--	--	--	--
Shoal Creek at W. 12th St., Austin (08156800)	Shoal	8,269	1.73	32.7	10.3	0	8.1	9.4	2.1	6.8	4.6	24.0	2.0
Boggy Creek at US 183, Austin (08158050)	Boggy	8,488	1.17	29.8	5.7	.3	6.7	4.8	9.2	3.3	7.2	22.4	10.5
Williamson Creek at Manchaca Rd., Austin (08158930)	Williamson	19,264	36.2	33.7	8.0	5.6	3.1	3.9	.4	1.1	7.1	15.0	22.1

Data for PACs in bed sediment are available for some Austin water bodies and for lakes and reservoirs around the country. Concentrations of PACs in LVSS samples are compared to bed-sediment samples collected from six sites in Town Lake in Austin during 1991–2004 and to bed-sediment samples representing sediment deposited from 1990 to about 2000 in 35 (metals) or 38 (organochlorine compounds and PAHs) lakes and reservoirs across the United States.

This study represents one of the most comprehensive uses of LVSS sampling to investigate concentrations, loads, and yields of metals, organochlorine compounds, and PAHs. In appendix 1, the advantages and disadvantages of LVSS sampling are discussed and compared to those of whole-water sampling as well as other approaches for investigating PACs, and LVSS results for metals and organochlorines in Shoal Creek are compared to those obtained by whole-water analysis for 1978–96. In appendix 2, LVSS results for metals are compared to those obtained by whole-water analysis for 1998–2004 at all of the sites investigated here.

Site Selection

Sampling sites were chosen in the watersheds of four Austin creeks: Barton, Shoal, Boggy, and Williamson. Barton and Williamson Creeks flow across primarily limestone and dolomite formations (Dillon and others, 1986). Shoal Creek flows across terrace deposits composed of gravel, sand, silt, clay, and limestone. Boggy Creek flows across terrace and alluvial deposits composed of gravels, sand, silt, clay, limestone, and chalk. All four creeks flow only intermittently. Base flow in these creeks, when it occurs, is typified by low SSC (for example, less than 10 milligrams per liter [mg/L]; Gandara and others, 1997, p. 100–102). Two of the creeks—Barton and Shoal—flow directly into Town Lake, and the watersheds of each are the largest of any of the local tributaries to Town Lake. The Main Spring outlet at Barton Springs also was sampled. The sampling sites are located at existing USGS streamflow-gaging stations (fig. 1; table 1).

Two sampling sites were chosen on Barton Creek (fig. 1). The first, where the creek is intersected by State Highway 71 (Barton 71), is just downstream from the confluence of Barton Creek and a major tributary, Little Barton Creek. The site has been used for many years by the USGS as a reference site; that is, it is assumed to be minimally affected by urban development. However, development of the watershed upstream from this site is increasing, as is traffic on State Highway 71—from 1990 to 2004 traffic volume on State Highway 71 almost tripled from 10,000 to 28,000 vehicles per day (Capital Area Metropolitan Planning Organization, 2005). As of 2003, the entire Barton Creek watershed was about 31-percent developed, and the watershed upstream from this site was about 28-percent developed. The largest category of developed land is residential (23 percent), of which about one-half is large-lot residential. For the purposes of this report, Barton 71 is assumed to represent a non-urban site. The second site, located just upstream from the dam separating Barton Springs pool from Barton Creek and the bypass routing water from Barton Creek around Barton Springs pool (Barton Above), represents the downstream end of the watershed. Urban development in the downstream part of the watershed is much denser than at Barton 71. As of 2000, about 67 percent of the watershed from Barton Above upstream to Barton Creek and Loop 360 was developed

Sources of Particle-Associated (Hydrophobic) Contaminants

Three major groups of PACs are commonly evaluated in aquatic systems: trace elements (metals), organochlorine compounds, and PAHs. The metals of primary concern in aquatic systems are cadmium, chromium, copper, lead, mercury, nickel, and zinc. Included with the metals here is the metalloid arsenic, whose behavior is similar to that of the metals. There are numerous natural and anthropogenic sources of metals to the environment. Metals occur in all geologic settings at different concentrations. In addition to natural sources, anthropogenic sources such as nonpoint-source pollution (road and parking lot runoff, landscape runoff, and faulty septic and wastewater systems), landfills, smelting, and industrial processes have released metals into the atmosphere and water. In the urban environment, some common sources of metals are rubber tires (zinc), pressure-treated wood (arsenic, chromium, copper), brake pads (copper), algaecides (copper), and lubricating oil and grease (lead, nickel, zinc).

The sale of many organochlorine compounds has been banned, but because of their persistence, they are still widespread in the environment (U.S. Environmental Protection Agency, 2005). The most commonly detected organochlorines are DDT and its breakdown products DDE and DDD, PCBs, chlordane, and dieldrin. DDT use was widespread in the United States during the 1950s and 1960s. Use peaked in the early 1960s and was banned in 1972 because of concern for its toxicity and persistence in the environment. PCB use peaked about 1970 and was banned in 1977 because of similar concerns. PCBs were widely used in products as diverse as vacuum pumps, compressors, and electrical equipment; they were used as a lubricant, a dielectric fluid, or a heat-transfer fluid. Chlordane use in agriculture, which was primarily for corn, was banned in 1978; however, its use for termite control was permitted until at least 1988. In addition, use of existing stocks by homeowners was permitted after 1988 and was found to be common in a 1990 survey (Whitmore and others, 1992). Agricultural use of dieldrin and its parent compound, aldrin, peaked in 1966 and was banned in 1974; its use on termites was banned in May 1987. Concentrations of these organochlorine compounds in sediment cores from the Nation's lakes and reservoirs, including a sediment core from Town Lake (Van Metre and Mahler, 1999), generally peak in the core intervals corresponding to the period of peak use; concentrations then decrease in more recently deposited sediment, except for chlordane and dieldrin (Van Metre and Mahler, 2005). In Town Lake, total DDT and PCBs decreased by about 90 and 70 percent, respectively, from peak concentrations in the 1960s to the late 1990s when the core was collected. Chlordane peaked in the early 1970s but had decreased only about 30 percent by the late 1990s. Dieldrin was not detected in the Town Lake core (P.C. Van Metre, U.S. Geological Survey, unpub. data, 1998).

PAHs represent the largest class of suspected carcinogens (Björseth and Ramdahl, 1985). They consist of two to seven fused benzene rings with hydrogen atoms attached to each carbon atom on the outside of the rings. Parent PAHs, those with only hydrogen atoms attached, can be alkylated by replacing one or more of the hydrogen atoms with alkyl groups (a carbon atom and three hydrogen atoms). In the environment, the presence and distribution of PAHs is largely a result of the incomplete combustion of petroleum, oil, coal, and wood (Edwards, 1983). The principal anthropogenic sources to the environment are thought to be vehicles, heating and power plants, industrial processes, and refuse and open burning (Sims and Overcash, 1983), although recently it has been demonstrated that parking lot sealcoat might be a major source to urban water bodies (Mahler and others, 2005). In a national study of trends by the U.S. Geological Survey, concentrations of PAHs in sediment cores from lakes and reservoirs increased significantly from 1970 to about 2000 in the majority of urban lakes studied (Van Metre and Mahler, 2005).

with a mix of single-family and multifamily residential, streets and roads, office, and commercial development (Bio-West, Inc., 2001). The drainage area from Barton Above upstream to Barton Creek and Loop 360 is 9 square miles (mi^2), whereas the entire watershed upstream from Barton Above is 125 mi^2. In this report, Barton Above is referred to as an urban site.

The sampling site on Shoal Creek (fig. 1) is near the confluence of the creek and Town Lake and is assumed to integrate sediment from the whole watershed. The Shoal Creek watershed is about 93-percent developed (2003 land-use data), with about 43-percent residential and about 23-percent commercial, office, and civic (Robert Botto, City of Austin, written commun., 2003). In this report, Shoal is referred to as an urban site.

The sampling site on Boggy Creek (fig. 1) was chosen because the Boggy Creek watershed is densely urbanized and an important riparian area for East Austin. The watershed includes the Robert Mueller Municipal Airport (closed in 1999) and part of the Interstate 35 commercial corridor. Boggy Creek is channelized for much of its length upstream from the sampling site. The Boggy Creek watershed is about 82-percent developed (2003 land-use data), with about 30-percent single-family residential and the remainder mostly transportation (Robert Botto, City of Austin, written commun., 2003). Industrial land use in the Boggy Creek watershed (9 percent) is about 4 times that in the Shoal Creek watershed, which has the next highest percentage. In this report, Boggy is referred to as an urban site.

Sediment-Quality Guidelines

Government agencies and researchers have developed sediment-quality guidelines (SQGs) to evaluate the toxicity of three groups of sediment-associated contaminants—metals, organochlorine compounds, and PAHs—to benthic biota. The SQGs are not regulatory limits, but they are a useful measure to evaluate the potential toxicity of bottom sediment to aquatic biota. Because the SQGs were developed for bottom sediment, and not sediment in suspension, they cannot be used to evaluate whether the suspended sediment collected by the study described in this report is, in itself, toxic to benthic biota. However, as stormwater flows recede, suspended sediment is deposited on streambeds and in downstream lakes, reservoirs, and estuaries. Van Metre and Mahler (2004), using the LVSS sampling method in a study comparing the chemistry of sediment deposited in reservoirs to that of suspended sediment from influent streams, determined that the concentrations of metals in the reservoir sediment and the suspended sediment were similar. They also found similarity in some cases, although not all, for organochlorine compounds and PAHs; when there were differences, the contaminant concentrations associated with suspended sediment were higher in most cases. Therefore, comparison of contaminant concentrations associated with suspended-sediment samples collected in this study to SQGs was deemed to be a reasonable approach for evaluation at least of relative contamination levels.

SQGs generally use two threshold levels to indicate levels of concern. The lower threshold is the concentration below which adverse effects to biota are not expected to occur; the upper threshold is the concentration above which adverse effects are expected to occur. In an attempt to resolve differences between various sets of guidelines, MacDonald and others (2000) developed a set of consensus-based SQGs, which are a numerical consensus of as many as six published sets of guidelines developed using similar methods. Statistical testing of the consensus-based SQGs showed them to be reliable predictors of observed toxicity and lack of toxicity to biota in bed sediment from a variety of settings in North America. The list of contaminants with SQGs (shown at right) is an indication of which hydrophobic contaminants are of most concern in lakes and streams. The two consensus-based SQG thresholds developed by MacDonald and others (2000) for each compound—the threshold effect concentration (TEC), below which adverse effects are not expected to occur, and the probable effect concentration (PEC), above which adverse effects are expected to occur—are used in this report for evaluation and comparison of sediment.

Metals	Organochlorine compounds	Polycyclic aromatic hydrocarbons (PAHs)
Arsenic	Dieldrin	Anthracene
Cadmium	Chlordane	Fluorene
Chromium	DDD	Naphthalene
Copper	DDE	Phenanthrene
Mercury	DDT	Benz[*a*]anthracene
Nickel	Total DDT	Benzo[*a*]pyrene
Lead	Total PCBs	Chrysene
Zinc		Fluoranthene
		Pyrene
		ΣPAH_{SQG}[1]

[1] Total PAHs used for consensus-based sediment-quality guidelines (Ingersoll and others, 2000).

The sampling site on Williamson Creek (fig. 1) was added to the list of sites after the first year of sampling because Williamson Creek contributes recharge to Barton Springs (Hauwert and others, 1998; Slade and others, 1986), and because after Barton Creek, Williamson is the next closest creek to Barton Springs. Of the six major creeks contributing recharge to Barton Springs, Williamson Creek has the most densely developed watershed at 65 percent; most of the development is single-family residential (34 percent) and transportation (15 percent) (Robert Botto, City of Austin, written commun., 2003). In this report, Williamson is referred to as an urban site.

The area contributing to Barton Springs comprises the watersheds of Barton, Williamson, Slaughter, Bear, and Onion Creeks to the west of the recharge zone (defined as the outcrop of the Edwards and Georgetown Limestones) and the recharge zone itself. Developed land use over the entire area was about 22 percent in 2000 (Bio-West, Inc., 2001), most of which was residential. For the purposes of this report, land use in the Barton Springs watershed is assumed to represent mostly non-urban uses.

Acknowledgments

We thank David Johns, Edward Peacock, Leila Gosselink, and Martha Turner of the City of Austin, and Robert Breault and William Selbig of the USGS for their helpful comments on the manuscript.

Methods of Sampling and Analysis

Approach

The general approach for LVSS sampling was to collect a sufficient amount of suspended sediment to allow analysis of the particulates (as opposed to the whole water) for concentrations of PACs on a mass-per-mass basis. This was accomplished by collecting large volumes of water and filtering out the suspended sediment. The suspended sediment then was analyzed for a suite of PACs.

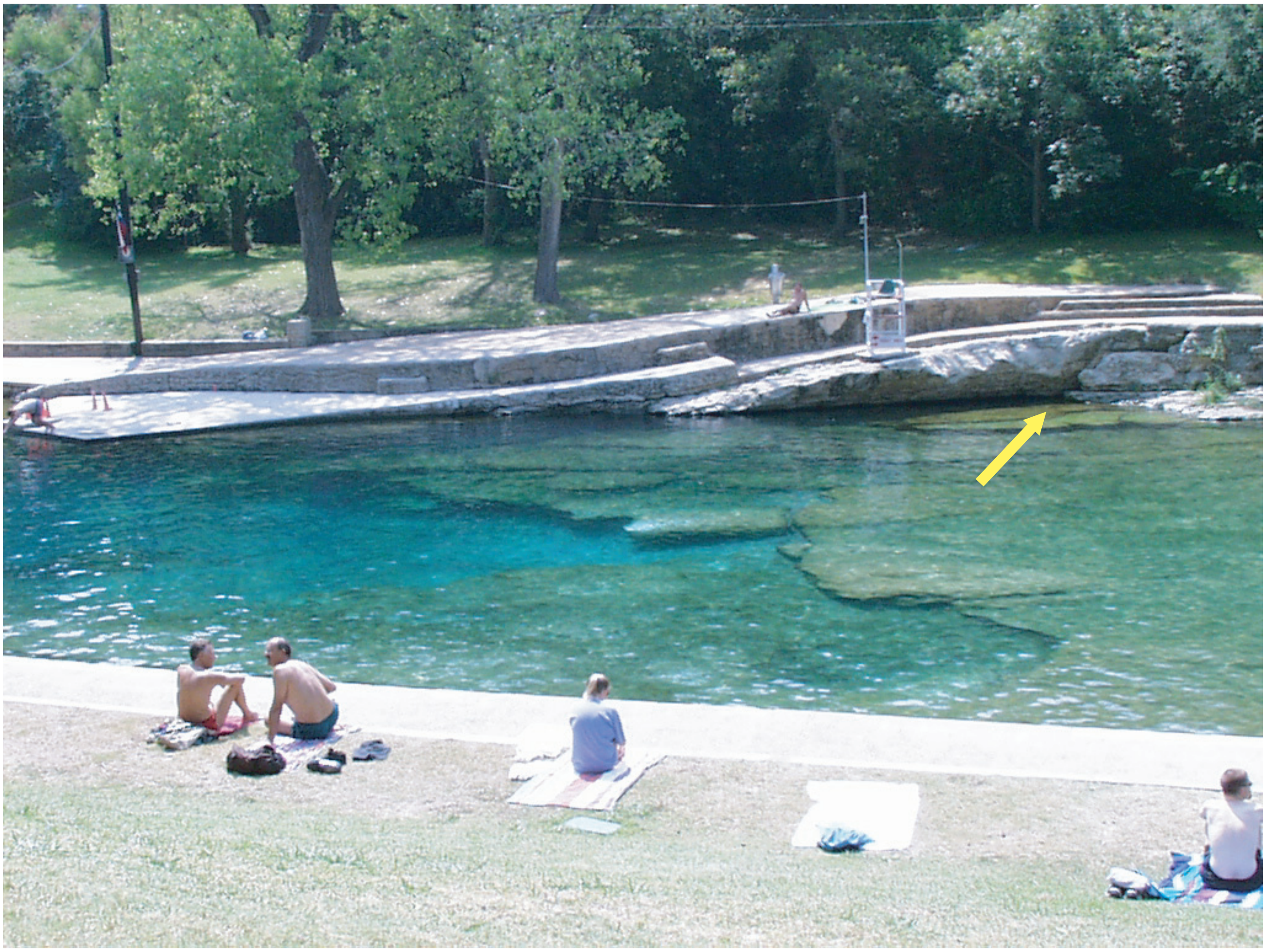


Figure 2. Site of sample collection at Barton Springs pool. The fissure from which samples were collected is indicated by the yellow arrow. (Photograph courtesy of David Johns, City of Austin.)

Flow-weighted composite samples were collected at all sites except Barton Springs. The constituent concentration in the composite sample is assumed to represent the event-mean concentration. During selected rain events at Shoal and Boggy Creeks discrete samples were collected over the event hydrograph and analyzed separately to investigate changes in sediment geochemistry during the event. The discrete samples consisted of either individual samples or a flow-weighted composite of two or more individual samples combined to give the best coverage of the hydrograph. The discrete samples also were used to compute event-mean concentrations of contaminants for each rain event for comparison to the other sites where only composite samples were analyzed. The event-mean concentration was computed by weighting the concentrations measured in the discrete samples by the proportion of the flow represented, then summing. For this application nondetections were counted as zero, thus for those events with one or more nondetections, the value reported represents a lower bound on the concentration.

Samples were collected from the fault orifice of Main Spring, the largest of the springs comprising Barton Springs (fig. 2). Water discharging from this orifice, which is located just upstream from the Barton Springs pool diving board, is assumed to be representative of water discharging from Main Spring, and it is from this orifice that the USGS and City of Austin collect routine water samples. For this study, suspended sediment was collected and isolated on-site by filtering continuously for several hours during the period of highest turbidity, usually 12 to 24 hours after rainfall.

Samples were collected during and after events of at least 0.25 inch (in.) of rainfall and following at least 10 days of no rainfall greater than 0.1 in. or rainfall that did not result in streamflow. The rainfall had to be sufficient to result in a creek stage that was high enough to trigger the automatic sampler and that remained high enough for five or more of the bottles in the autosampler to fill. If at that time the stage had not dropped to near that of base flow, additional samples were collected, either manually or by replacing the containers in the autosampler with clean containers.

Event loads and yields of contaminants were computed for each of the events sampled. The load of each contaminant for the event was computed as

$$L = C \times SSC \times V \times (1 \times 10^{-9}), \qquad (1)$$

where

L = event load of contaminant (metals in grams, organic contaminants in milligrams);

C = contaminant concentration in storm-composite sample (metals in milligrams per kilogram, organic contaminants in micrograms per kilogram);

SSC = suspended-sediment concentration for the event-composite sample (milligrams per liter);

V = total volume of water passing by streamflow-gaging station for the period sampled (liters); and

1×10^{-9} = unit conversion factor.

The event yield, the mass of contaminant per unit area of watershed for the event, was obtained by dividing the event load, in grams (metals) or milligrams (organics), by the watershed area, in acres (reported for metals in milligrams per acre, organics in micrograms per acre).

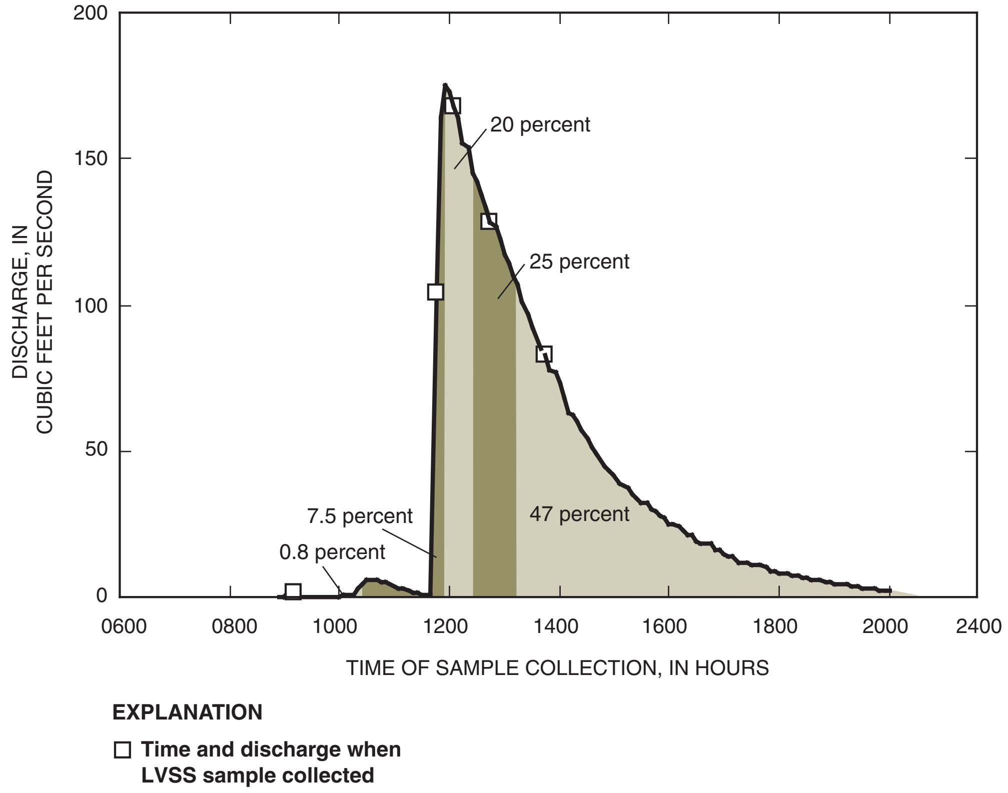
Sample Collection, Compositing, and Processing

With few exceptions, samples of creek water for suspended-sediment analysis were pumped into seven 9-liter (L) high-density polyethylene carboys using an Isco 6700 water sampler. In some cases the carboys were lined with polytetrafluoroethylene (PTFE) bags if other constituents such as dissolved pesticides were to be analyzed from the same sample. Sampler intake lines are 3/8-in. diameter and made of either polyvinyl chloride or PTFE depending on the water-quality constituents to be analyzed. Stream channel geometry, debris, and flow velocity were considered when placing the sampler intakes. The sampler intake hose was positioned as close to the channel center as possible. The intakes were positioned as low as possible to enable collection of a first-flush sample. In some cases under high base-flow conditions, sampler intakes were raised. As a result, intake placement generally was from 5.5 to 18 in. above the creek bed. The water sampler was programmed to collect the samples after a prescribed rise in stage and then at set time intervals after that (Wilde and others, 1999).

The appropriate sample-collection time interval for each site was chosen on the basis of historical streamflow data, with the objective of covering the typical event hydrograph. On occasion, if stage was still high after the autosampler had completed its seven-bottle carousel, additional samples were collected to give better hydrograph coverage. These samples were collected either by refilling the autosampler carousel with clean bottles and restarting the sampler, or by lowering a carboy through the water column using a rope, similar to the method described by Wilde and others (1999), at the estimated centroid of flow at the approximate location of the autosampler intake. The samples were returned to the USGS Texas Water Science Center laboratory in Austin and kept chilled until compositing, which usually took place within 24 hours of sample collection.

Samples were composited on the basis of volume of flow represented by each discrete sample measured at the gaging station (Fisk and others, 1994) (fig. 3). The volume of flow was computed from 15-minute stage data recorded at each gaging station and the stage-discharge relation. Stage-discharge relations have been developed for each of the sites and are updated regularly by measurements made with current meters (Gandara and others, 1997). The first discrete sample was assumed to represent the time interval from when the stage began to rise to halfway between collection of the first and second discrete samples. The second and following samples were each assumed to represent the time interval from halfway between collection of the previous sample and the sample of interest to halfway between collection of the sample of interest and the following sample. The last sample was assumed to represent the time interval from halfway between collection of the next to the last sample and the last sample to the time when the stage was interpreted to have returned to base flow or to have begun a new rise in response to new rainfall. Each discrete sample was agitated or churned in a high-density polyethylene or PTFE churn. A volume of water proportional to the volume of flow represented by the discrete sample was either poured off (if agitated) or removed through the churn spigot (Horowitz and others, 1994) and placed with the other samples in a PTFE (20-L) or high-density polyethylene (50-L) container. Agitation was used at the beginning of the project, but after the first 2 years the churn was used instead to avoid negatively biasing particle-size distribution.

Suspended-sediment sample for analysis was obtained by separating the sediment from the water by inline filtration, as described in Mahler and Van Metre (2003). To obtain the 3 to 5 grams of sediment necessary for analysis of organochlorine compounds and PAHs, water was pumped from the stirred, composited sample either through a pre-baked 293-millimeter (mm) diameter, 0.7-micrometer (μm) pore-size glass-fiber filter (GFF) (samples collected before November 1, 2000) or through a 293-mm diameter, 0.45-μm pore-size PTFE filter (samples collected after November 1, 2000) held in a stainless steel (SS) plate filter holder. The flow-weighted composite was mixed in a high-density polyethylene churn splitter, and the sample was churned during filtration at a rate of about 9 inches per second. Two to three GFFs were used to obtain the amount of sediment required for each sample for analysis of organic compounds. Following filtration, the filters were placed (together) in a pre-baked glass jar, chilled, and shipped to the USGS National Water Quality Laboratory (NWQL), where they were extracted together. PTFE filters had to be pre-wetted with methanol to allow the initial passage of water because PTFE is very hydrophobic. Water was pumped until the filter clogged. The loaded



Note: Final composited sample volume made up of corresponding discrete samples in proportions shown.

Figure 3. Discharge hydrograph showing associated large-volume suspended-sediment (LVSS) discrete sample collection times and proportion of total volume of flow represented by each discrete sample for rain event at Shoal Creek, April 26, 1999.

filter was placed in a locking plastic bag, and the sediment was removed from the filter by massaging gently. For a single sample the same PTFE filter was used several (usually 3) times. The resulting sediment-water slurry was poured into a baked glass vial (any sediment clinging to the bag was rinsed into the vial with a jet of deionized, distilled water), chilled, and submitted to the NWQL for analysis. The volume of sample filtered was measured and recorded. Two or more 250-milliliter (mL) samples of the unfiltered water collected periodically during churning were submitted to the USGS Louisiana Water Science Center sediment laboratory in Baton Rouge, La., for determination of SSC and, in most cases, the silt-sand break (proportion of material, by weight, with an equivalent particle-size diameter exceeding 63 µm) by gravimetric analysis. For samples filtered with GFFs, the reported concentrations of organic compounds (in mass of organic contaminant per liter of water filtered) were converted to mass of organic contaminant per mass of sediment by dividing the mass per volume concentration by the mean SSC (mass contaminant/volume water x volume water/mass sediment = mass contaminant/mass sediment). For samples filtered with PTFE filters, the NWQL determined the mass of sediment analyzed, and the concentrations were reported as mass of organic contaminant per mass of sediment.

To obtain samples for major element and metal analyses, water was pumped through a PTFE filter held in an acrylic filter holder. The 142-mm diameter, 0.45-µm pore-size PTFE filter was sprayed with methanol to allow the initial passage of water. Water was pumped until the filter clogged; then the filter was placed in a locking plastic bag, and the sediment was removed from the filter by massaging gently. The resulting water-sediment slurry was freeze-dried and ground before analysis of major elements and metals. Results were reported as mass of element per mass of sediment.

At Barton Springs, samples were filtered on-site. Continuous filtration was necessary because the sediment concentration was too low for a flow-weighted composite sample to be practical. The samples from Barton Springs thus represent the sediment discharging during the period of highest turbidity. Turbidity was monitored with a handheld turbidimeter. Filtering began when the turbidity had increased to about 7 nephelometric turbidity units and continued until sufficient sediment for analysis had been obtained. This usually required from 6 to 8 hours, and in almost every case turbidity peaked during the sampling period. Samples were collected through tubing inserted into a fracture extending to the south side of the pool from the Main Spring opening (fig. 2). Water was pumped continuously either through the organic contaminants filtration apparatus or the metals filtration apparatus, described above, until the filters were clogged. Once on the filters, samples were processed as described above. Samples for SSC analysis were collected approximately hourly during sampling.

Rainfall Data

Rainfall data used for this study are from the City of Austin Flood Early Warning System rain-gage network (City of Austin, 2001). The proportion of each watershed represented by each gage was determined by the Theissen polygon method (Chow and others, 1988). The number of rain gages used in each watershed was four (Barton 71), nine (Barton Above), six (Shoal and Williamson), and three (Boggy). An areal-weighted average rainfall rate for 1-hour intervals was computed for each watershed for each event. Total rainfall (inches) was computed by summing the 1-hour area-weighted rainfall. Average intensity (inches per hour) was computed by dividing the total rainfall by the storm duration. The duration was chosen to be the period between the first and last rainfall intensities of 0.1 inch per hour (in/hr) or more. If rainfall intensity did not exceed 0.1 in/hr, the average intensity was taken as the maximum hourly intensity.

The antecedent rainfall period was computed as the number of days between the sampled event and the preceding total rainfall of 0.1 in. or more. The antecedent flow period was taken as the number of days between the sampled event and the preceding rainfall sufficient to cause an increase of 1 cubic foot per second (ft^3/s) daily average flow in the creek (5 ft^3/s daily average flow for Barton Springs). The antecedent rainfall and antecedent flow periods are not necessarily the same, because under some conditions rainfall infiltrates into the soil without producing any streamflow.

Analytical Methods

At the NWQL, organochlorine compounds and PAHs in sediment were measured in organic-solvent extracts using a dual capillary-column gas chromatograph with dual electron capture detectors. The method uses a Soxhlet extraction with dichloromethane followed by gel permeation cleanup and adsorption chromatographic fractionation. The extraction and cleanup procedures follow those of Foreman and others (1995) and Furlong and others (1996) with modifications. Modifications include a Florisil column cleanup step for organochlorine compounds and quantification following the procedures of Wershaw and others (1987) and quantification of PAHs by selected ion monitoring (SIM) (E.T. Furlong, U.S. Geological Survey, oral commun., 2000). These modifications were incorporated as new methods and published by the USGS in 2003 (Noriega and others, 2003; Olson and others, 2004). The organochlorines analyzed are aldrin, chlordane, dieldrin, *p,p'*-DDD (DDD), *p,p'*-DDE (DDE), *p,p'*-DDT (DDT), endosulfan, endrin, heptachlor, heptachlor epoxide, lindane, methoxychlor, mirex, toxaphene, and PCBs (Aroclors 1242, 1254, and 1260). Nineteen parent PAHs, 10 specific alkyl-PAHs, and 27 alkyl-PAH homologues were determined for this study. Total PAH was computed as the sum of these, excluding perylene. Also reported in this study is total PAH used for the consensus-based sediment-quality guidelines (ΣPAH_{SQG}), computed as the sum of naphthalene, 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, benz[*a*]anthracene, dibenz[*a,h*]anthracene, benzo[*a*]pyrene, chrysene, fluoranthene, fluorene, phenanthrene, and pyrene (Ingersoll and others, 2000).

Reporting levels for the organochlorine analyses of samples collected by LVSS vary primarily for two reasons. First, the laboratory reporting level (LRL) is affected by the amount of sample analyzed. The contaminant mass quantified is based on the peak size in the chromatogram, and the area under the peak is normalized by mass or volume of sample to obtain a concentration. A sample containing only a very small mass of sediment will result in a chromatogram in which the peaks are proportionately smaller than those for a sample with the same contaminant concentration but a greater mass of sediment; the smaller the peak, the more difficult the accurate determination of the concentration and the higher the LRL. Second, concentrations on the GFF filters are reported by the NWQL as mass of contaminant per volume of water filtered; the actual mass of sediment analyzed is not known by the laboratory because it remains adhered to the GFF. Because the mass-per-mass concentration is computed by dividing the mass-per-volume concentration by the SSC, two samples with the same LRL in units of mass per volume will have different LRLs in units of mass per mass unless their SSCs are identical.

Concentrations reported by the NWQL for organics analyses are flagged as “estimated” (indicated by an E where reported) when they are qualitatively identified as present, but the reported concentrations have a greater uncertainty than usual (Childress and others, 1999). Situations that result in an “estimated” flag include concentrations that are less than the established LRL but still detectable in the analyst’s judgment, samples for which surrogate recovery was not within control limits, and samples from sample sets for which recovery from the set spike was not within control limits. Estimated concentrations were included in computations of event loads and yields and for statistical analyses.

Elemental analyses were done at the USGS Mineral Resource Program Analytical Laboratories in Denver, Colo. The freeze-dried sediment was completely digested using a mixture of hydrochloric-nitric-perchloric-hydrofluoric acids (Briggs and Meier, 2002). This method is referred to as a “total digestion.” Elemental concentrations were determined on concentrated-acid digests using inductively-coupled plasma mass spectrometry (Briggs and Meier, 2002); concentrations of mercury were determined by cold-vapor atomic adsorption (Fishman and Friedman, 1989). The organic carbon concentration was determined if sufficient sample was available. Organic carbon was computed as the difference between total carbon, analyzed by oxidation of carbon to carbon dioxide through combustion in an oxygen atmosphere (Curry, 1996), and total carbonate carbon, analyzed by coulometric titration (Papp and others, 1996). The major elements analyzed are aluminum (Al), calcium (Ca), iron (Fe), potassium (K), magnesium (Mg), sodium (Na), phosphorus (P), and titanium (Ti). The metals analyzed and reported here are arsenic (As) (a metalloid), cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), nickel (Ni), lead (Pb), and zinc (Zn).

Quality Assurance of Chemical Data

The purpose of quality assurance (QA) in a water-quality study is to demonstrate the level of reliability and precision of the data produced. Four types of QA data are presented in this section: laboratory, routine field, and two methods-development tests.

Laboratory Data

Samples submitted to the USGS Mineral Resource Program Analytical Laboratories and the USGS NWQL are grouped into sample sets. For elemental analyses at the USGS Mineral Resource Program Analytical Laboratories, a sample set consists of 20 environmental samples, a duplicate of one of those environmental samples, three blank samples, and four standard reference materials (SRMs). For organochlorine and PAH analyses at the NWQL, a sample set consists of 12 environmental samples, a duplicate from those samples, a blank sample, a spiked sample (the blank with known amounts of target analytes added), and a Certified Reference Material (CRM). Thus four QA samples are analyzed with every 12 environmental samples; a set of 12 environmental samples might include from 1 to 12 samples for this study. Additionally, surrogate compounds are added to each environmental sample before extraction and analysis. A surrogate is a compound that is not a target analyte but behaves chemically like one or more of the target analytes. Surrogates are quantified and reported with each environmental sample and are an indication of the effectiveness of recovery of target analytes.

Laboratory QA results for major elements and metals analyses associated with this study were acceptable (Horowitz and others, 1994). For major element and metals analyses, 72 percent of the 1,263 analyses of SRMs were within 10 percent of the published concentrations. Low concentrations of major elements and metals were detected in some blanks and ranged from 5 to 10 percent of the minimum value detected in any environmental sample. Concentrations of environmental samples were not corrected but reported as the raw data.

Laboratory QA results for organochlorine and PAH analyses associated with this study also were generally acceptable. None of the organochlorines were detected in any of the 19 laboratory blanks. Ninety-three percent (255 of 273) of organochlorine surrogate recoveries were within NWQL-established control limits. Ninety-five percent (146 of 154) of organochlorine spike recoveries were within control limits. When spiked values did not fall within control limits and the analyte was detected in the environmental sample, the laboratory flagged the concentration in the environmental sample as estimated. Ninety-three percent (127 of 137) of blank and spike surrogate recoveries were within NWQL control limits. Ninety percent (65 of 72) of organochlorine CRMs were within NWQL control limits. For PAHs, 90 percent (202 of 225) of the analyses had no detectable concentration in the laboratory blank. Ninety-seven percent (144 of 149) of the CRM results were within control limits. One hundred percent of laboratory spike recoveries were within control limits.

Routine Field Data

Field QA consisted of two parts: evaluation of the representativeness of samples collected by autosampler, and evaluation of the collection and processing methods for reproducibility.

For this study, samples collected by autosampler were assumed to be representative of cross-sectional flow. To test this assumption, SSC in samples collected by the autosampler were compared to those in samples collected manually using USGS protocols on three different occasions. The manual samples were collected using an isokinetic sampler; in the first instance, a single vertical sample was collected, and in the second and third, depth-integrated samples were collected using the equal-width-increment (EWI) method. The methods are detailed in Webb and others (1999) and the results discussed below.

At Barton 71, a non-urban site, discrete samples were collected simultaneously with the autosampler and manually 5 times over the hydrograph of an event on November 1, 1998. Each manual sample was a single vertical depth-integrated sample collected with a DH–9 suspended-sediment/water-quality sampler at the same site where the autosampler intake was positioned. The samples were analyzed for SSC, and the relative percent difference (RPD, defined as the difference between a pair of results divided by their average) between SSC for the two types of samples ranged from -24.2 to 16.5 percent; the average RPD was 3.7 percent, with the average SSC from the autosampler samples slightly higher than from samples obtained using the depth-integrated method. One of the five manually collected samples contained 2-percent sand; the four other manually collected samples and all five autosampler samples contained no sand.

At Shoal, one of the most urban sites, samples collected with the autosampler and manually were compared for two rain events, one on December 23, 2002, and one on June 4, 2003. In both instances, the manual samples were collected by a team of four technicians spaced evenly across the width of the channel (EWI sample). Each technician collected a depth-integrated vertical sample by lowering a 3-L frame sampler into the water,

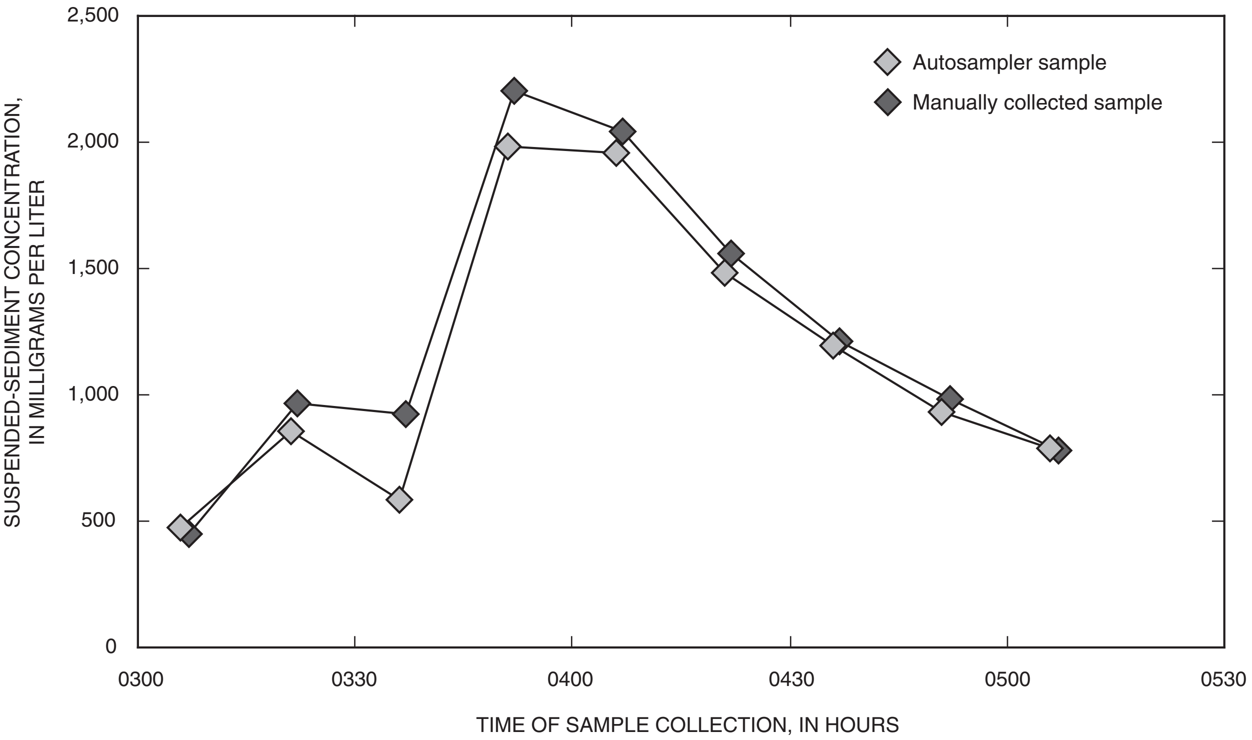


Figure 4. Comparison of suspended-sediment concentrations in discrete samples collected by autosampler and manually during a rain event at Shoal Creek, June 4, 2003.

and the four samples were combined. One of the drawbacks of this approach is that the strength of the current diverts the sampler from a true vertical trajectory. Under such conditions, the intake nozzle of the frame sampler may jam into, or disturb, the bed sediment, resulting in a sample that is biased toward a higher SSC (Michael Canova, U.S. Geological Survey, oral commun., 2005).

For the December 23, 2002, event at Shoal, nine samples were collected simultaneously over the hydrograph with the autosampler and manually with the EWI method. Each set of nine samples was combined into a flow-weighted composite sample, and the flow-weighted composite was analyzed for a suite of water-quality constituents, including SSC. The discrete samples were not analyzed. The RPD between SSC for the two types of samples was 7.3 percent, with the SSC from the autosampler samples higher than that in the manually collected samples (2,442 mg/L and 2,269 mg/L, respectively). For the June 4, 2003, event at Shoal, nine discrete samples were collected simultaneously over the hydrograph by each of the two methods, but for this event the discrete samples were analyzed for SSC (fig. 4). The average RPD was 4.3 percent; in this case the manually collected samples tended to have a higher SSC than those collected with the autosampler. These samples might be positively biased for the reasons mentioned in the previous paragraph. The SSC in one of the nine manually collected samples exceeded, by 22 percent, the SSC in the sample collected by the autosampler; it is possible that the manually collected sampler contained sediment inadvertently picked up from the streambed (Michael Canova, U.S. Geological Survey, oral commun., 2005).

On the basis of the overall agreement between samples collected manually (either single vertical, depth-integrated or EWI method) and samples collected with the autosampler, the autosampler samples were judged to be sufficiently representative of flow through the cross section. This was true for both a rural and an urban site; therefore, it was assumed to be true for the other sites included in this study.

For suspended sediment, field QA consisted mostly of duplicate samples. Field blanks cannot be run because there is no way to process “clean” sediment in the field, as normally would be done with a water sample by processing laboratory-purified blank water in the field. However, an unused baked GFF was digested after 5 and 16.6 L of distilled water were filtered through it to determine if the filter contained a detectable mass of any organochlorine compounds or PAHs. The mass of PAH detected in the blank GFFs was, in all cases, about an order of magnitude higher in the filter that had less water pumped through it, indicating that some PAH leaches from the GFFs. The mass obtained from the GFF that had 5 L pumped through it ranged from 7.0×10^{-5} microgram (µg) for phenanthrene to 9.8×10^{-4} µg for anthracene. The importance of the mass of PAH contained in the filters can be put into perspective by comparing it to the mass of PAH collected at Barton Springs, where some of the lowest concentrations of PAHs occurred. In sediment discharging from Barton Springs during the May 26, 1999, event, when the lowest PAH concentration occurred, the mass of PAH detected ranged from 1.5 to 20 times that in the GFF through which 5 L was pumped (except for anthracene). Because hundreds of liters of water usually are pumped through the filter at Barton Springs during the filtration process, the likelihood that PAHs from the filter are contributing to PAHs measured at Barton Springs is small. PAH concentrations from other sites are higher than those from Barton Springs; thus although the GFFs contain a small amount of detectable PAH, it is expected to contribute little to the overall PAH concentrations detected in the samples.

Duplicate environmental samples were analyzed for major elements, metals, organochlorines, and PAHs. To collect a duplicate sample, two samples were processed from the same churn, thus the duplicate is evaluating the reproducibility of the sample processing and laboratory analysis, but not the water collection from the stream. The inorganics analyses require about one-tenth of the mass of sediment required for the organics analyses, so it was possible to submit more duplicate samples for determination of major element and metals concentrations than for determination of organochlorine and PAH concentrations, for which sediment mass often was a limiting factor. RPD was computed for each pair (one environmental and one duplicate) of samples. Fifteen pairs of duplicate samples were analyzed for major elements and metals. For major elements, 81 percent (95 of 118) of the pairs had an RPD of less than 5 percent, and all had an RPD of less than 10 percent. For metals, 81 percent (83 of 103) of the pairs had an RPD of less than 5 percent, and 88 percent had an RPD of less than 10 percent. The best agreement was for Zn (all RPDs less than 5 percent), and the worst agreement was for Cu (67 percent of pairs had RPDs less than 5 percent; 73 percent had RPDs less than 10 percent). Two pairs of duplicate samples were analyzed for organochlorines. In both pairs, all organochlorines were less than the LRL, except for DDE in one sample. The RPD for that sample was 18 percent, which is good considering that the concentrations were extremely low (0.83 and 1.0 microgram per kilogram [μg/kg]) and reported as estimates. For PAHs, the median RPD was 11.3 percent; 44 percent (12 of 27) of pairs had an RPD less than 10 percent, and 78 percent (21 of 27) of pairs had an RPD less than 20 percent. Analysis of duplicate samples using the same methods for a similar study in Fort Worth, Tex., resulted in a median RPD of 8.5 percent for organochlorine compounds and 6.5 percent for PAHs (Van Metre and others, 2003).

Methods-Development Test Data

Two methods-development tests were carried out. The first was to determine if the large SS filter support used for the organics sample could be used to collect the sample for metals analysis without causing metals contamination. Use of the larger filter would simplify the collection process and allow a larger mass of sample to be collected at Barton Springs. The second test was to determine if a PTFE filter could be used in the place of the GFF for the organics analysis, thus eliminating the need to include the GFF in the extraction process and the use of SSC to convert from mass per volume filtered to mass per mass.

The SS filter support was used to process a metals sample at Barton Springs after a rain event. The results included an anomalously high Zn concentration (1,820 milligrams per kilogram [mg/kg]). A duplicate of the sample yielded the same results. To determine if Zn or other metals might be leaching out of the SS filter holder, samples from Shoal and Boggy were split; the two splits were filtered using the SS filter holder and the acrylic filter holder. For most of the metals, the difference in concentrations among the samples was less than 10 percent, within the margin of analytical error, and the sample with the higher concentration was not consistently the SS sample. The metal with the most consistent difference greater than 10 percent, As, was 12 and 16 percent higher in both SS samples. Cr, Cu, and particularly Ni concentrations were greatly elevated in the SS sample from Boggy, but not in the SS sample from Shoal. Concentrations of As, Cr, Cu, and Ni did not appear to be elevated in the Barton Springs sample relative to other samples from that site. Zn was slightly elevated in the SS samples from both Shoal and Boggy but by less than 10 percent. However, because the Barton Springs sample with the anomalously high Zn concentration was filtered using the SS filter support, the results from that sample were not reported with the other Barton Springs samples or used in the statistical analyses. It was decided to continue use of the acrylic filter holder to avoid any chance of contamination by the SS filter holder.

The second methods-development test was to determine if a large PTFE filter could be used in the SS filter holder instead of the GFF to obtain sediment samples for organics analyses. There are two advantages to this substitution. First, the laboratory can determine the mass of sediment being analyzed instead of computing it from the volume of water filtered and the SSC. Second, the sediment can be removed from the PTFE filter and digested like any other sediment sample; for samples filtered with a GFF, the entire filter must be digested along with the sediment, which creates potential analytical interferences. NWQL personnel had reported that the GFF extracts often contained an emulsion that was difficult to remove, which resulted in a chromatogram baseline that was difficult to resolve.

Three samples were split to do the test. For each sample, one split was filtered through two GFFs, and the other split was filtered through a PTFE filter. The GFFs for the first split were submitted together to the laboratory for extraction and analysis; for the second split, the sediment was removed from the PTFE filter and submitted to the laboratory for extraction and analysis of organochlorine compounds and PAHs. For organochlorines, in one case an analyte was detected in the GFF sample but not in the PTFE sample (dieldrin), and in two cases an analyte was detected in the PTFE sample but not the GFF sample (DDT and PCB 1242). In all cases where an analyte was detected in both samples, the concentration detected using the PTFE filter exceeded by 1.3 to 3.6 times the concentration detected using the GFF filter. For PAHs, in all cases the concentration detected using the PTFE filter was equal to or exceeded that detected using the GFF. The median ratio between the two results (PTFE/GFF) was 2.0, and except for naphthalene in one of the three samples, the ratios ranged from 1.0 to 2.8. These results indicate that interferences caused by the co-extraction of the sediment and GFF might result in underestimation of the organochlorine and PAH concentrations. In contrast, when the PTFE filter is used, the sediment is removed from the filter before analysis and only the material of interest (the sediment) is extracted. Thus it appears that use of the PTFE filters produces results superior to those using the GFF filter, and results from the samples filtered with the GFF filters should be regarded as minimum values. Therefore all samples collected after November 1, 2000, were processed using PTFE filters.

Sample Concentrations, Loads, and Yields

Samples included in this report were collected between March 12, 1999, and June 9, 2004. The number of rain events sampled at each site during this period ranged from 8 to 12 (fig. 5). Organic carbon content, grain-size information, and concentrations, loads, and yields of suspended sediment, major elements and metals, organochlorine compounds, and PAHs for all sites and all events are given in appendixes 3–5. Concentrations express how contaminated the sediment is on a per-mass basis. Event loads represent the total mass of contaminant transported over an event. Event yields define the mass of contaminant produced per unit area of watershed for an event.

Sample Concentrations

Concentrations, reported as the mass of contaminant per mass of sediment, are a direct measure of sediment contamination. Concentrations can be compared between sites and events to assess relative contamination. They also can be compared to SQGs developed by regulatory agencies and researchers, as described earlier in the "Introduction" section, to determine relative degree of contamination (although not toxicity).

Major Elements

The relative abundance of major elements in a sediment indicates the minerals present (appendix 3.1). For example, high Al and Fe concentrations occur in clay minerals and their Fe-oxide coatings, whereas high Ca and Mg concentrations are present in limestone and dolomite. Differences in abundances of major elements in the samples might reflect differences among the geology of the watersheds, flow regime of the creeks, and even land use.

Major element assemblages varied relatively little at individual sites, because they likely reflect the geology of the upstream watershed, which is unchanging on the time scale considered here. The median coefficient of variation (C_V), a measure of variability defined as the standard deviation divided by the mean, for the major elements at the creek sites was less than 0.25 in all cases. At Barton Springs the median C_V was 0.31; the greater variation might result from a mix of autochthonous sediment from the aquifer and allochthonous sediment from four different watersheds. At the four creek sites with urbanized watersheds (Barton Above, Shoal, Boggy, and Williamson), the greatest variation was for either Na or P, two elements that have anthropogenic sources.

The highest mean P concentrations occurred at Barton Above, Shoal, and Williamson. This enrichment could reflect greater use of fertilizers or greater septic and wastewater releases in these watersheds than in the others, or it could reflect geologic or other conditions. At Barton Springs, the mean Ca concentration was lower than at the creek sites, and the mean Fe and Mg concentrations were higher. Lower Ca concentrations probably result from the dissolution of calcite particulates during transport through the aquifer or from dilution by autochthonous clays. Higher Fe concentrations reflect the contribution of Fe-rich autochthonous clays. The abundance of Mg (about 3.5 times more than at the creek sites) probably results from flow conduits that feed Barton Springs intersecting with strata of easily mobilized microcrystalline dolomite particles, which can compose as much as 40 percent, by weight, of Barton Springs suspended sediment (Mahler and Lynch, 1999, p. 171).

Trace Elements (Metals)

Rocks and soils contain a wide variety of metals in addition to the major elements discussed above. Eight of the metals that have important anthropogenic sources and that are toxic to biota at high concentrations in sediment have established SQGs: As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn. Concentrations of these eight metals for the events and sites sampled are shown in figure 6 and listed in appendix 3.1.

In general, concentrations of metals were low in comparison to SQGs (fig. 6). The PEC—the concentration above which adverse effects to benthic biota are expected—was exceeded only 4 times: twice for Zn, once for Ni, and once for Pb, although the TEC—the concentration below which no adverse effects to benthic biota are expected—was exceeded in 216 of 461 concentrations, just less than one-half. On the basis of comparison to the SQGs, Shoal and Williamson were the sites with the highest concentrations of metals. Seventy-two percent of the individual metal analyses from Shoal were above the TEC and two analyses were above the PEC (Pb and Zn). Sixty-seven percent of the individual metal analyses from Williamson were above the TEC, and two analyses were above the PEC (Ni and Zn). Barton 71 was the least contaminated site with respect to metals, with only 15 percent of samples exceeding the TEC. For all sites combined, the frequency of a metal sample exceeding the TEC was in the order of Cr (81 percent) > Ni (79 percent) > As (74 percent) > Zn (65 percent) > Pb (48 percent) > Cu (14 percent) > Cd (12 percent) > Hg (0; no samples exceeded the TEC).

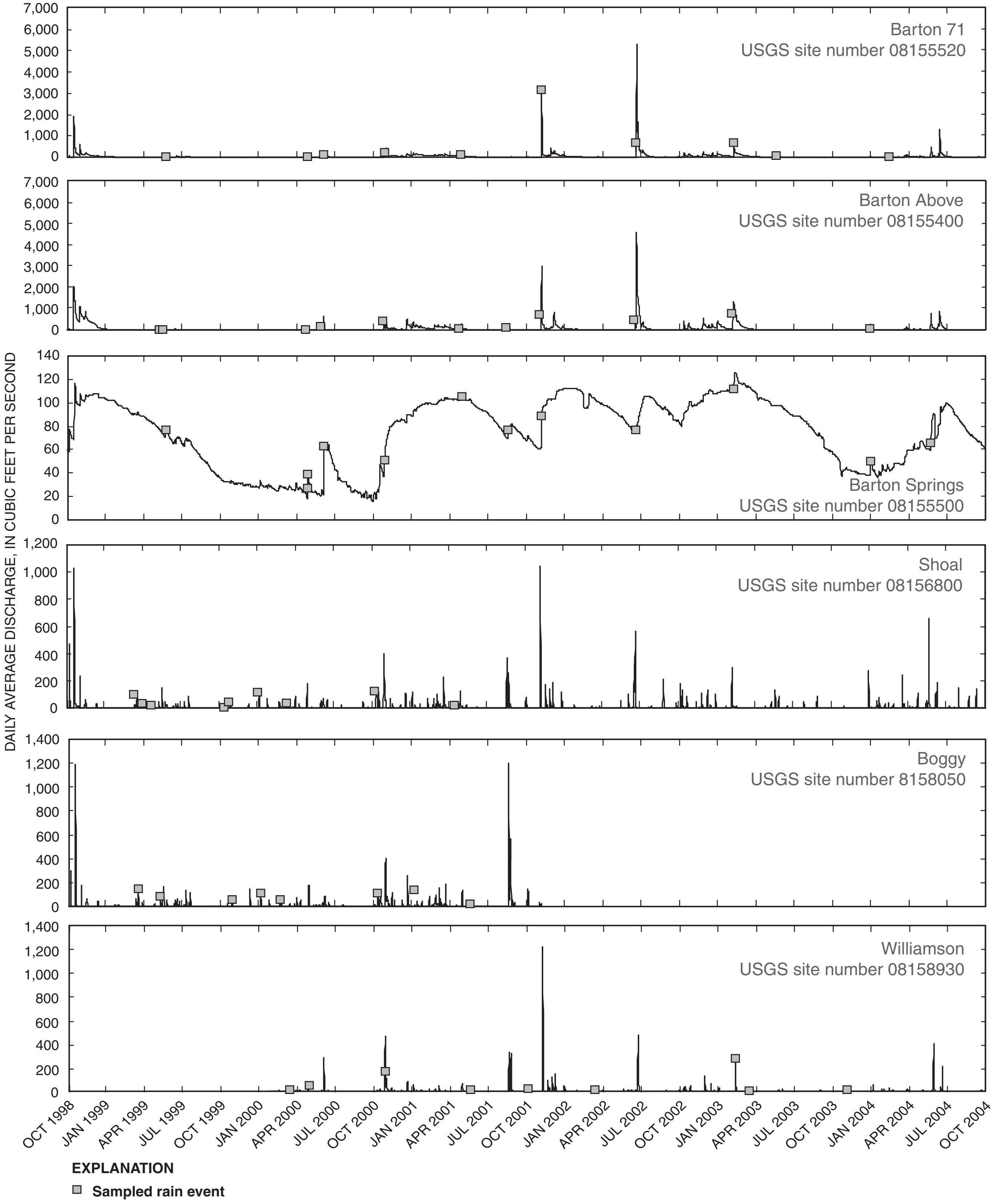


Figure 5. Discharge hydrographs (1998–2004) showing sampled rain events (1999–2004) in urban creeks, Austin, Texas.

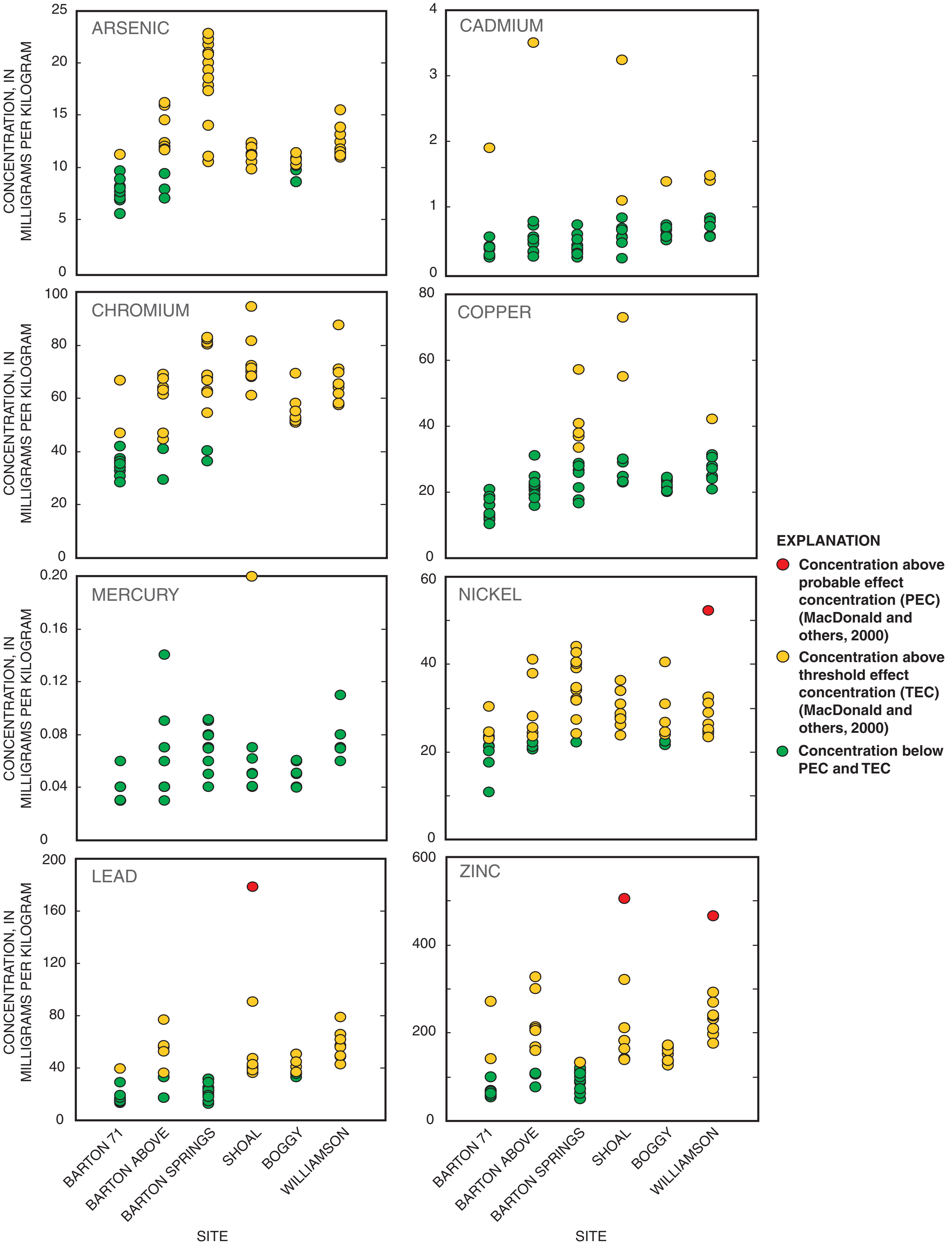


Figure 6. Concentrations of metals in suspended sediment in urban creeks, Austin, Texas, 1999–2004.

Although almost one-half of the metals samples had a concentration above the TEC (appendix 3.1), for some metals at some sites, these concentrations might reflect natural background conditions. Metals occur over a wide range of concentrations in geologic materials and sometimes can exceed SQGs (particularly the TEC) even when there is no anthropogenic source. One way to determine whether a metal has a natural or anthropogenic source is to compare the metals concentrations of a suite of sediment samples from a similar geologic setting to Fe concentrations (Harwell and others, 2003; Mahler, 2003; Menounou and Presley, 2003). If the concentration of a metal varies proportionately with Fe, which is assumed to have a natural source, the metal also is assumed to have a natural source. When metals concentrations are graphed in relation to Fe concentrations, the samples for which the metals concentrations vary proportionately with Fe concentrations will plot along a straight line; samples that plot above the straight line have a disproportionately high concentration of the metal in comparison to Fe.

Six of the eight metals (As, Cr, Cu, Ni, Pb, and Zn) show a statistically significant linear relation with Fe at Barton Springs (fig. 7; table 2), indicating a probable natural mineralogic origin for these metals at this site. Statistical significance was tested using the linear correlation coefficient (Pearson's r), which measures the linear association between two variables (Helsel and Hirsh, 1995), and which varies from 0 (no correlation) to ±1 (perfect correlation). The coefficient of determination (r^2, the square of Pearson's r) indicates the fraction of variation in the metals concentration explained by its relation to Fe (Helsel and Hirsch, 1995). The relations between metals concentrations and Fe concentrations in most samples of suspended sediment at Barton Springs indicate that metals at Barton Springs likely have a natural mineralogic source and might be used to estimate natural concentrations of metals in samples from other sites on the basis of their Fe concentrations. In contrast, none or only one of the metals is significantly correlated with Fe for three of the urban sites (Shoal, Boggy, and Williamson) (fig. 7; table 2).

Concentrations of Cu and Pb in suspended sediment correlate with Fe only at Barton Springs. Concentrations of Pb from the four urban sites and concentrations of Cu and Zn in some samples from Shoal and Williamson plot as a "cloud" above the linear relation defined by other sites, indicating potential anthropogenic sources for these two metals in some samples at these sites (fig. 7). The distance between the Fe regression line indicating expected background concentrations and the highest plotted points indicates that concentrations of these three metals in some samples are several times higher than the background concentrations. Elevated concentrations of Cu, Pb, and Zn in storm runoff in urban areas, particularly roadway runoff, are well documented (for example, Barrett and others, 1995; Christensen and Guinn, 1979; Davis and others, 2001); it is not unexpected that these three metals are elevated in suspended sediment in Austin urbanized watersheds. In contrast, concentrations of Cu and Pb at Barton 71 mostly plot along or below the linear relation described for Barton Springs, indicating that Cu and Pb in most samples from this site likely have a natural mineralogic source.

Relations between Fe and As and Cr indicate natural mineralogic sources for most samples: Most plot either along or below the linear relations described by samples from one or more sites. The indication of a mineralogic source for As at Barton Springs (fig. 7) is of particular note because As concentrations are higher at this site than at any of the creek sites.

The correlation approach was less successful for identifying potential anthropogenic sources for Cd and Hg. There were no significant relations between Fe and Cd at any site. This might indicate an anthropogenic source for Cd in some samples, but it also might indicate the tendency of Cd to be less well associated with iron oxides than the other seven metals considered here (Horowitz and Elrick, 1987). Similarly, Horowitz and Elrick (1987) found Hg to be less well correlated with Fe than As, Cr, Cu, Ni, Pb, or Zn.

In conclusion, Pb associated with suspended sediment at the urban sites appears to have an anthropogenic source. There is no indication of consistent anthropogenic sources for As, Cr, or Ni. The regression method used here is inconclusive for Cd and Hg.

Organochlorine Compounds

The organochlorine compounds detected at one or more of the sites are the insecticides dieldrin, chlordane, and DDT and its breakdown products, DDD and DDE, and the industrial compounds PCBs (fig. 8; appendix 3.2). Use of all of these compounds is banned or restricted in the United States. At least one of these compounds was detected at least once at every site (although LRLs varied widely as a result of the variable amount of sediment available for analysis) and in some cases exceeded the TEC and even the PEC.

Dieldrin was detected at Barton Above, Shoal, Boggy, and Williamson. It was detected most frequently at Boggy (three of eight samples) and Shoal (five of nine samples), but did not exceed the PEC in any sample. The highest concentrations were detected in samples from Shoal.

Contamination by chlordane is the most severe, on the basis of comparison of concentrations to SQGs (fig. 8). Chlordane was detected in suspended sediment at Barton Above, Shoal, Boggy, and Williamson; the highest concentrations were at Shoal and Williamson. In all cases where chlordane was detected, concentrations exceeded the TEC; more than one-half of the detections at Shoal, Boggy, and Williamson exceeded the PEC.

DDT or one of its breakdown products, or both, were detected at all sites, but these compounds exceeded the PEC less frequently than did chlordane. The PEC was exceeded only in samples from Boggy (once each for DDD and DDT and twice for DDE) and from Shoal (once for DDT and twice for DDE). Overall, the highest median concentrations of DDT and its breakdown products were detected in Boggy.

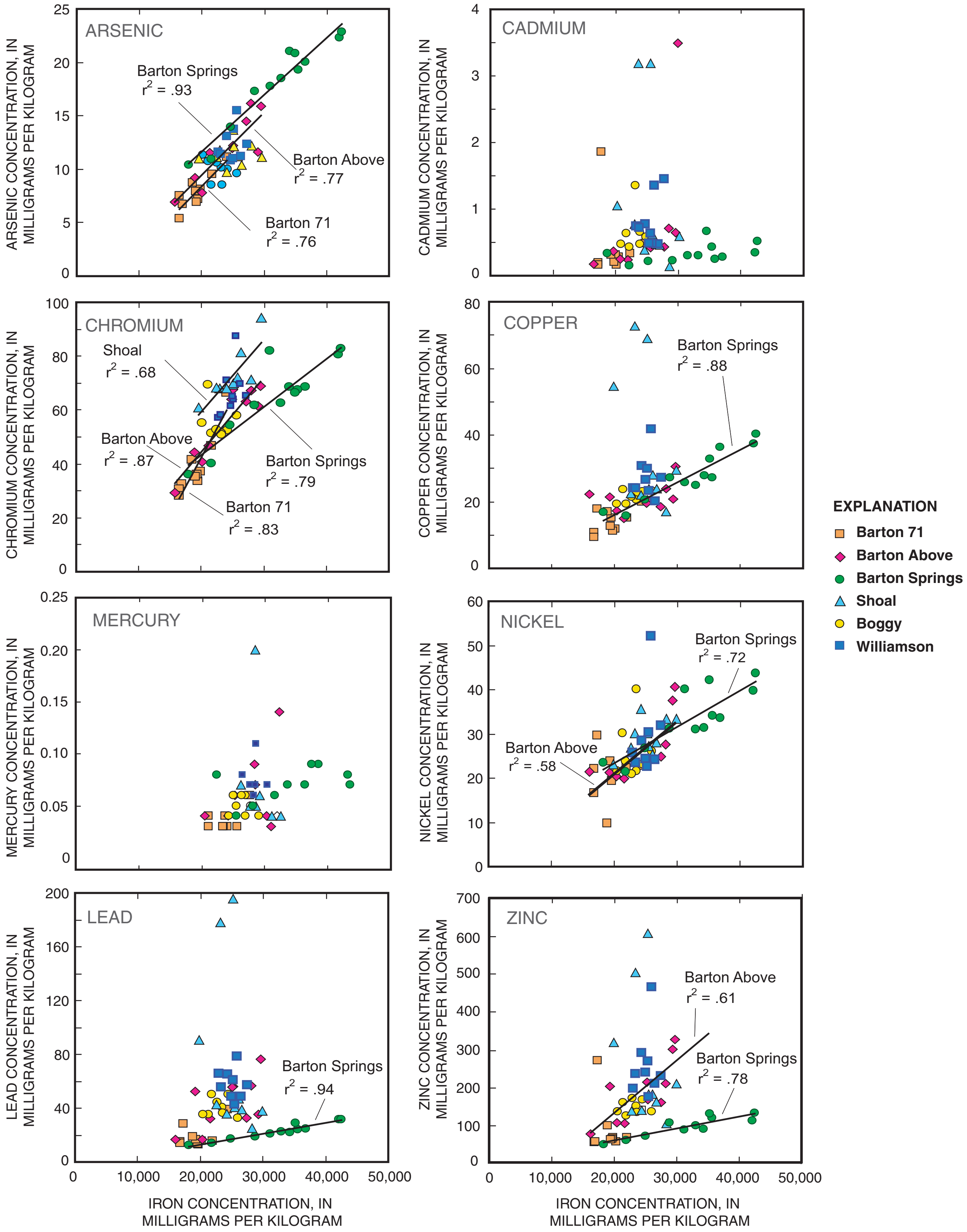


Figure 7. Relations between metals and iron concentrations in suspended sediment in urban creeks, Austin, Texas, 1999–2004, indicating natural and anthropogenic effects.

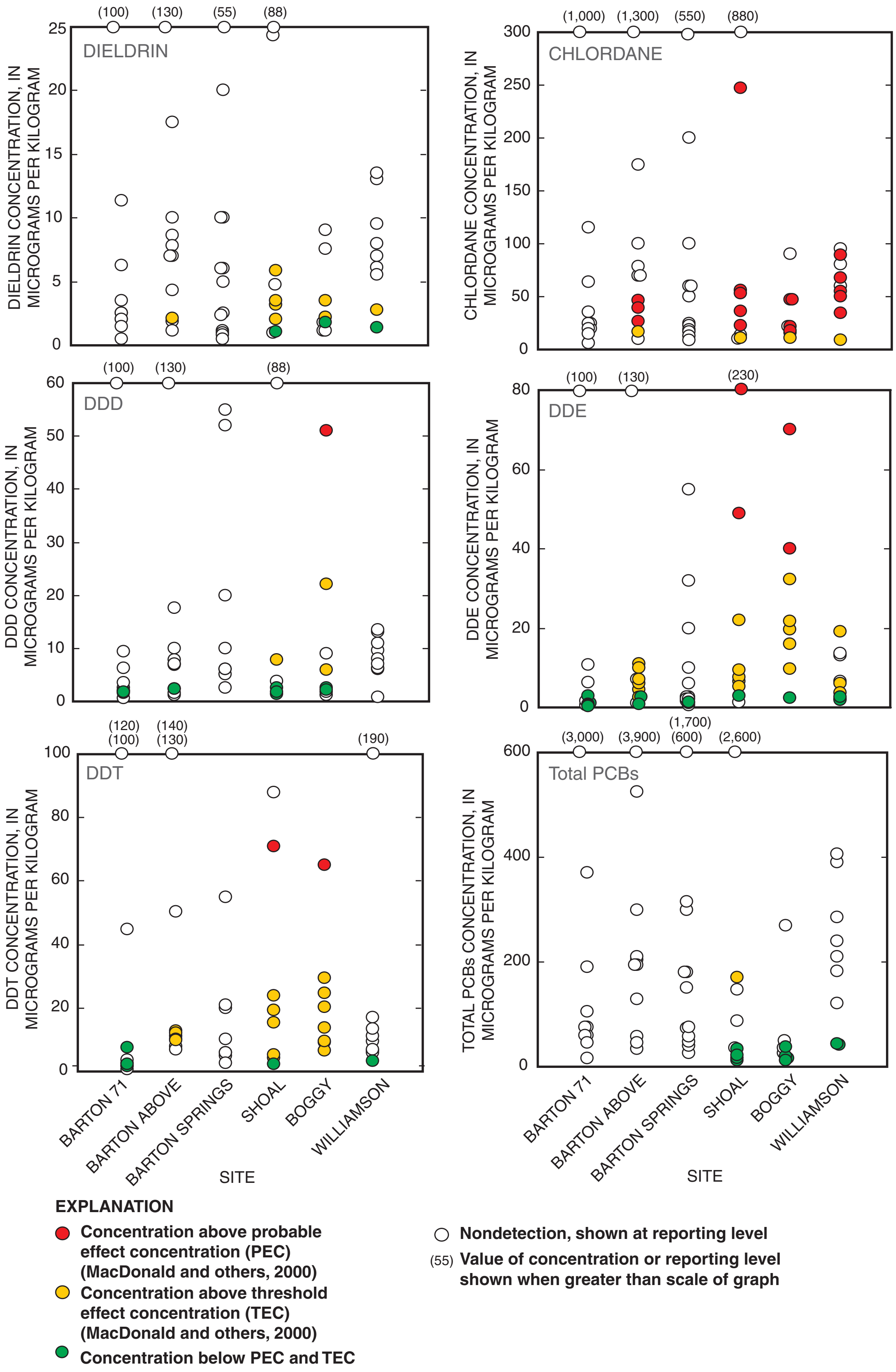


Figure 8. Concentrations of organochlorine compounds in suspended sediment in urban creeks, Austin, Texas, 1999–2004.

Table 2. Linear correlation coefficients (Pearson's r) between concentrations of metals and total polycyclic aromatic hydrocarbons used for consensus-based sediment-quality guidelines (ΣPAH_{SQG})[1] for urban creeks, Austin, Texas, 1999–2004.

[Significant correlation (p-value < .05) in bold]

Sampling site	Cadmium	Chromium	Copper	Mercury	Nickel	Lead	Zinc	Iron	ΣPAH_{SQG}
Barton 71									
Arsenic	-.11	**.89**	.59	.48	.01	.57	.06	**.87**	.21
Cadmium		-.02	.53	.58	.60	.54	**.96**	-.17	-.01
Chromium			**.74**	.74	-.01	**.73**	.16	**.91**	.13
Copper				.61	.11	**.80**	**.66**	.56	.36
Mercury					.02		.89	.56	.09
Nickel							.52	.07	-.52
Lead				.87	.35		.71	.50	.08
Zinc								-.04	.14
Barton Above									
Arsenic	.57	**.89**	.45	.36	.57	**.71**	.61	**.88**	.14
Cadmium		.47	**.83**	**.82**	**.79**	**.69**	**.71**	.55	.05
Chromium			.37	.41	.58	**.75**	**.70**	**.93**	-.01
Copper				.58	**.73**	**.68**	**.67**	.40	-.18
Mercury					.47		.54	.33	.64
Nickel							**.89**	**.76**	-.21
Lead				**.74**	.52		**.74**	.60	.32
Zinc								**.78**	.01
Barton Springs									
Arsenic	**.59**	**.90**	**.92**	.51	**.86**	**.94**	.48	**.97**	-.01
Cadmium		.43	.47	.28	.46	.46	.33	.48	.14
Chromium			**.83**	.40	**.91**	**.85**	.45	**.89**	-.18
Copper				.52	**.85**	**.94**	.31	**.94**	-.10
Mercury					.46		-.07	.51	-.33
Nickel							.56	**.85**	.14
Lead				.49	**.90**		.51	**.97**	.14
Zinc								.44	.25
Shoal									
Arsenic	.60	-.15	.49	-.70	-.07	.61	.63	.13	.16
Cadmium		-.27	**.94**	**.99**	-.04	**.99**	**.97**	-.31	.56
Chromium			-.32	-.28	.39	-.32	-.24	**.82**	-.58
Copper				**.96**	-.19	**.96**	**.97**	-.44	**.77**
Mercury					-.14		**.96**	-.28	.63
Nickel							-.09	.57	-.53
Lead				**.99**	-.09		**.99**	-.35	.66
Zinc								-.29	**.68**
Boggy									
Arsenic	.25	.44	.24	.31	-.53	-.27	.54	-.39	.13
Cadmium		-.06	.06	.42	-.10	.15	.64	-.02	**.76**
Chromium			.54	.24	.15	-.49	.21	-.24	.11
Copper				.38	.02	-.20	.58	.54	.07
Mercury					-.36		**.69**	-.21	.66
Nickel							-.30	.12	-.28
Lead				.37	-.40		.28	-.02	.49
Zinc								.08	.59
Williamson									
Arsenic	.54	**.76**	**.69**	.73	**.91**	.48	.65	.23	-.05
Cadmium		.51	**.64**	-.27	**.71**	.64	.58	.41	-.15
Chromium			**.80**	-.12	**.85**	.58	**.87**	.46	-.02
Copper				-.66	**.82**	**.81**	**.95**	.14	.14
Mercury					.45		-.74	-.04	.08
Nickel							**.82**	.33	-.02
Lead				-.48	.62		**.80**	-.17	.28
Zinc								.15	0

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[1] From Ingersoll and others (2000).

Total PCBs (defined as the sum of PCB Aroclors 1242, 1254, and 1260) were detected at Shoal, Boggy, and Williamson. They were detected infrequently at Williamson (two of nine samples) but more frequently at Shoal and Boggy (five of nine and four of eight samples, respectively); the highest concentration was at Shoal. No samples exceeded the PEC, but concentrations are similar to those in suspended sediment in Donna Canal, Hidalgo County, Tex., where concentrations of PCBs are elevated in fish and a ban on fish possession has been enacted (Mahler and others, 2002).

Polycyclic Aromatic Hydrocarbons

The PAHs singled out for discussion are those with a consensus SQG: $\Sigma\text{PAH}_{\text{SQG}}$, anthracene, fluorene, naphthalene, phenanthrene, benz[*a*]anthracene, benzo[*a*]pyrene, chrysene, fluoranthene, and pyrene (fig. 9; appendix 3.3). The highest concentrations were consistently detected at Williamson, with concentrations in some samples exceeding 10,000 µg/kg for many of the heavier molecular-weight PAHs, for example, benzo[*a*]pyrene, chrysene, fluoranthene, and pyrene. The mean $\Sigma\text{PAH}_{\text{SQG}}$ at Williamson was about 4 times higher than the next highest mean at Shoal. Also at Williamson, the PEC for the six PAHs with three or more rings (phenanthrene, benz[*a*]anthracene, benzo[*a*]pyrene, chrysene, fluoranthene, and pyrene) and for $\Sigma\text{PAH}_{\text{SQG}}$ was exceeded by eight of the nine samples.

PAH concentrations at Barton Above, Shoal, and Boggy were about 5 to 8 times lower than those at Williamson; however, $\Sigma\text{PAH}_{\text{SQG}}$ concentrations at Shoal exceeded the PEC in two of the nine samples. Concentrations of PAHs at Barton 71 and Barton Springs were 13 and 19 times lower than at Boggy, the site with the next-highest concentrations. The TEC was exceeded once each by four PAHs at Barton 71 and was equaled once by benzo[*a*]pyrene at Barton Springs.

PAH assemblages—that is, the relative proportions of different PAHs in any given sample—are an indication of PAH source. PAH compounds exist in the environment as assemblages of two to seven fused benzene rings, each of which might be present in both parent (only hydrogen atoms attached to the outside of the rings) and variously alkylated forms (alkyl groups replacing one or more of the hydrogen atoms), called alkyl homologues. Uncombusted (petrogenic) sources (oil seeps, petroleum spills, coal, natural organic matter) contain predominantly 2- and 3-ringed compounds and proportionally more alkyl homologues, whereas combustion (pyrogenic) sources (diesel exhaust, coal combustion, forest fires) contain predominantly 4- and 5-ringed parent compounds (Eganhouse and Gossett, 1991; Hites and others, 1981). Sediment at sites minimally affected by anthropogenic activities tend to have low PAH concentrations (less than 1,000 µg/kg) and ratios of the 2- and 3-ringed compounds to combustion compounds greater than 1.0 (U.S. Geological Survey, unpub. data, 1997–2005; Van Metre and Mahler, 2004; Van Metre and others, 2000). The 2- and 3-ringed PAHs include naphthalene, fluorene, phenanthrene, anthracene, and their homologues; the combustion compounds are fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, and benzo[*g,h,i*]perylene. The ratios of 2- and 3-ringed compounds to combustion compounds for samples from the different sites are shown in figure 10. Samples from Williamson (except for one outlier) show the lowest ratios, typical of combustion sources; these samples also have the highest PAH concentrations. Barton Above, Shoal, and Boggy have ratios that are slightly higher. The highest ratios, reflecting a more petrogenic (natural) source, are detected in sediment from Barton 71. Ratios intermediate between those at Barton 71 and the urban sites are detected in Barton Springs sediment, even though concentrations are similar to those at Barton 71. The ratios thus reflect the contribution of anthropogenic PAHs of a combustion origin to the suspended sediment at the more urbanized sites and indicate that, although concentrations of PAHs are relatively low at Barton Springs, some sources might be anthropogenic.

Recent investigations by the USGS and the City of Austin have indicated that parking lot sealcoat is a previously unrecognized but potentially dominant source of PAHs to local urban creeks (Mahler and others, 2005; Mahler and others, 2004). To determine whether some of the PAHs in suspended sediment might be coming from abraded parking lot sealcoat, three indicator ratios for coal tar were compared: fluoranthene/pyrene, indeno[1,2,3-*cd*]pyrene/benzo[*g,h,i*]perylene, and benzo[*a*]pyrene/benzo[*e*]pyrene. In graphs that combine these ratios, similarities and differences between parking lot sealcoat and samples from the different sites are apparent (fig. 11). In general, most of the samples from the creek sites with more urbanization in their immediate watershed have ratios that are similar to ratios in samples of washoff from parking lots sealed with coal-tar emulsion based sealcoat, which indicates that abraded sealcoat is a possible PAH source. Samples from Barton 71 plot farther from samples from the sealed parking lots, with mostly lower values of fluoranthene/pyrene and mostly higher values of benzo[*a*]pyrene/benzo[*e*]pyrene. Samples from Barton Springs have ratios dissimilar to those from the sealed parking lot samples, with lower values of indeno[1,2,3-*cd*]pyrene/benzo[*g,h,i*]perylene and fluoranthene/pyrene and a wider range in values of benzo[*a*]pyrene/benzo[*e*]pyrene. Although concentrations of PAHs at Barton 71 and Barton Springs are low, the PAH assemblages do not resemble each other. The graphs of ratios indicate that some proportion of the PAHs in some samples from Barton 71 might come from parking lot sealcoat, but sealcoat does not appear to be a source of PAHs in Barton Springs sediment. The difference between the PAH assemblage in Barton Springs and the other sites might occur because of the age of the sediment discharging from Barton Springs, which is unknown, and the fact that it is exposed to water for much, if not all, of its storage time in the aquifer.

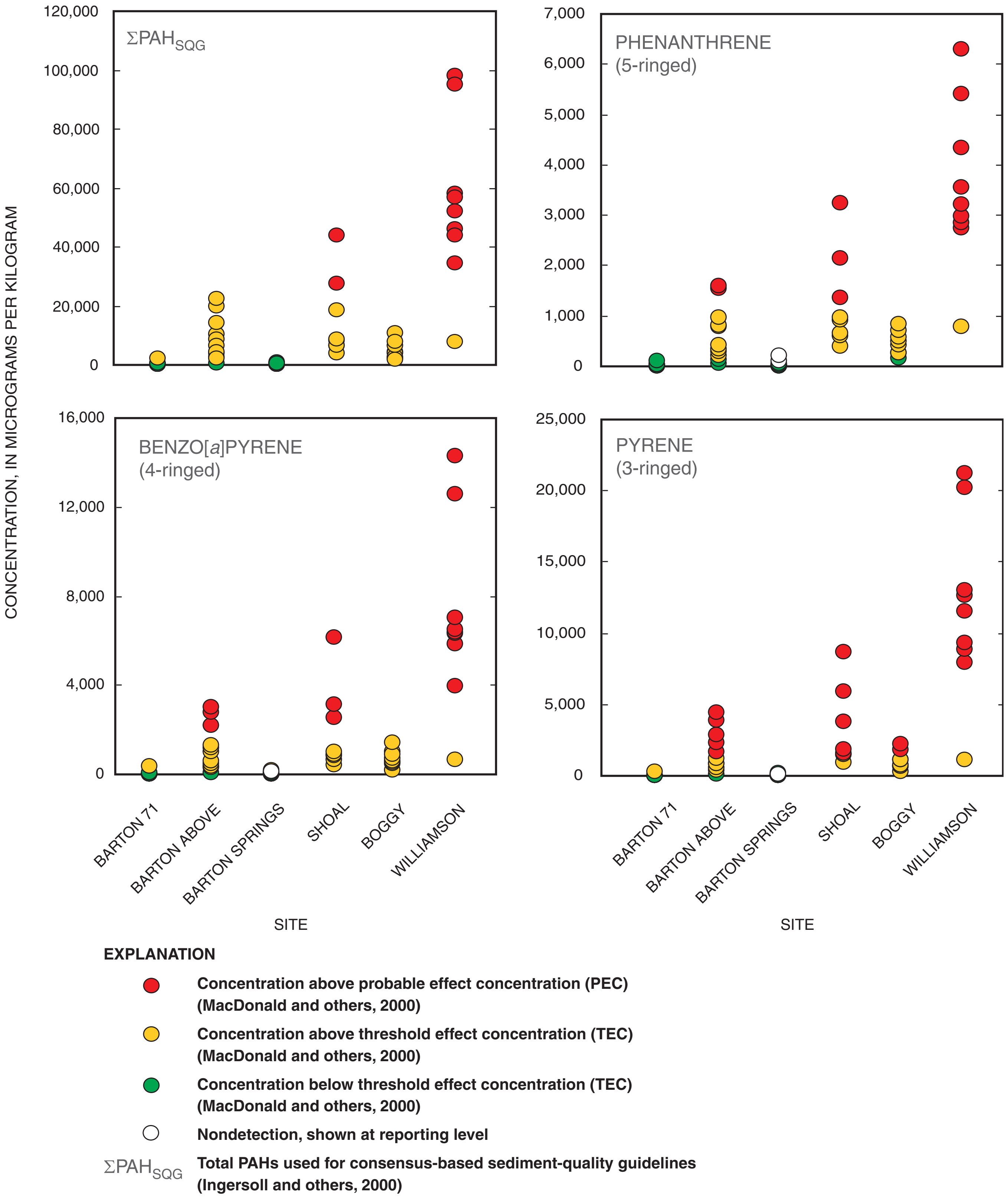
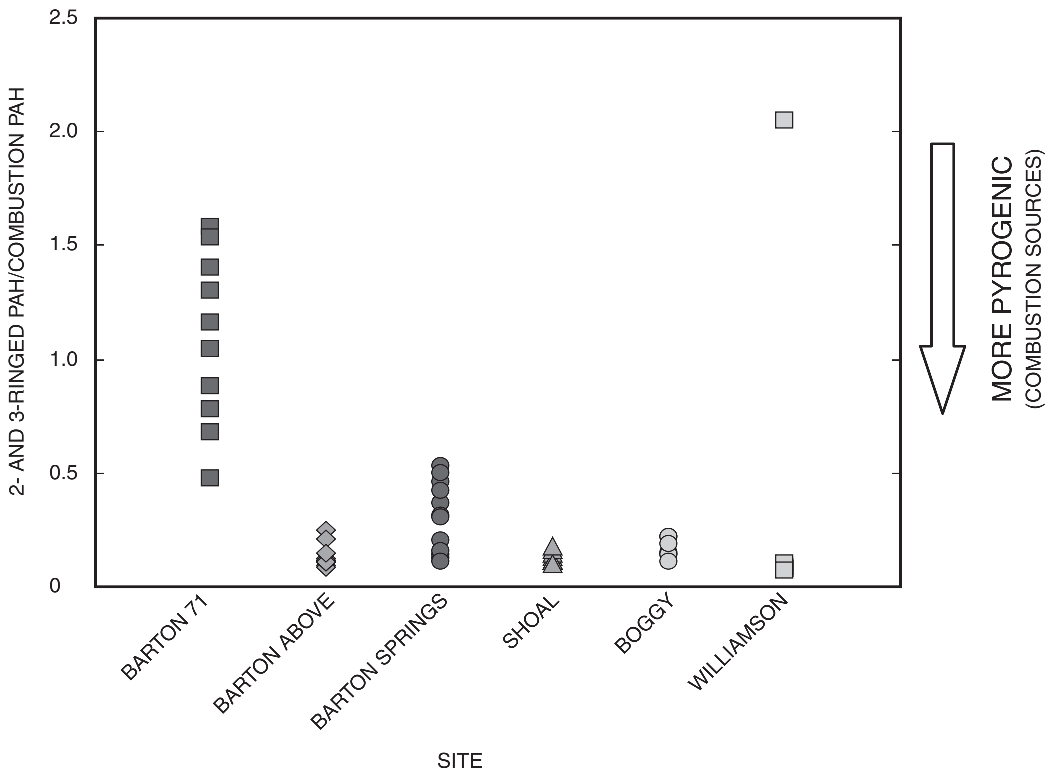


Figure 9. Concentrations of selected polycyclic aromatic hydrocarbons (PAHs) in suspended sediment in urban creeks, Austin, Texas, 1999–2004.



EXPLANATION

2- AND 3-RINGED PAH/ COMBUSTION PAH	**Ratio of sum of 2- and 3-ringed parent PAHs and their alkyl homologues to sum of 10 of the 4- and 5-ringed parent PAHs indicative of combustion sources**

Figure 10. Source-indicator ratio of polycyclic aromatic hydrocarbons (PAHs) in urban creeks, Austin, Texas, 1999–2004.

Variation in Concentrations

Different variations in concentration were evident for the three groups of contaminants (metals, organochlorine compounds, and PAHs). The difference in variation between contaminants and between sites can be compared using the C_v. For those datasets with censored data (nondetections), the mean and standard deviation were estimated using the robust probability plotting method; for datasets with as much as 60-percent censored data, estimates of the mean determined by this method have no more error than if there were no censoring (Helsel and Cohn, 1988).

SSCs by site had C_vs ranging from 0.53 to 1.0, with a median of 0.80. The variability was least at Boggy (C_v of 0.53) and greatest at Barton Above (C_v of 1.0).

Concentrations of metals showed the least variation. The C_vs computed for each metal at each site ranged from 0.08 (Cu at Boggy) to 1.24 (Cd at Barton Above), with a median (all metals and all sites combined) of 0.24. If the sites are considered individually, the median C_v for the eight metals ranged from 0.14 (Boggy) to 0.61 (Shoal). If the metals are considered individually, the median C_v ranged from 0.16 (As and Cr) to 0.70 (Cd).

Variability in the organochlorine compounds was about 3 times greater than for metals: the C_v for each compound at each site ranged from 0.15 to 1.94, with a median of 0.93 (all compounds and all sites combined). Considering the sites individually, the median C_v for organochlorine compounds ranged from 0.60 (Barton Above) to 1.16 (Shoal). Considering the compounds individually, dieldrin had the least variability (median C_v of 0.57), and total PCBs had the most variability (median C_v of 1.36); these two organochlorine compounds were detected mostly at only two sites (Shoal and Boggy).

Variability in the PAHs was similar to that for organochlorine compounds: by compound and by site, the range in C_v was greater (0.41 to 2.2), but the median was slightly less (0.88). The median C_v for PAHs at the different sites ranged from 0.52 (Williamson) to 1.5 (Barton 71). Naphthalene had the least variability (median C_v of 0.65) and fluorene had the greatest (median C_v of 0.99).

Concentrations of metals probably have less variability than concentrations of organochlorines, PAHs, and suspended

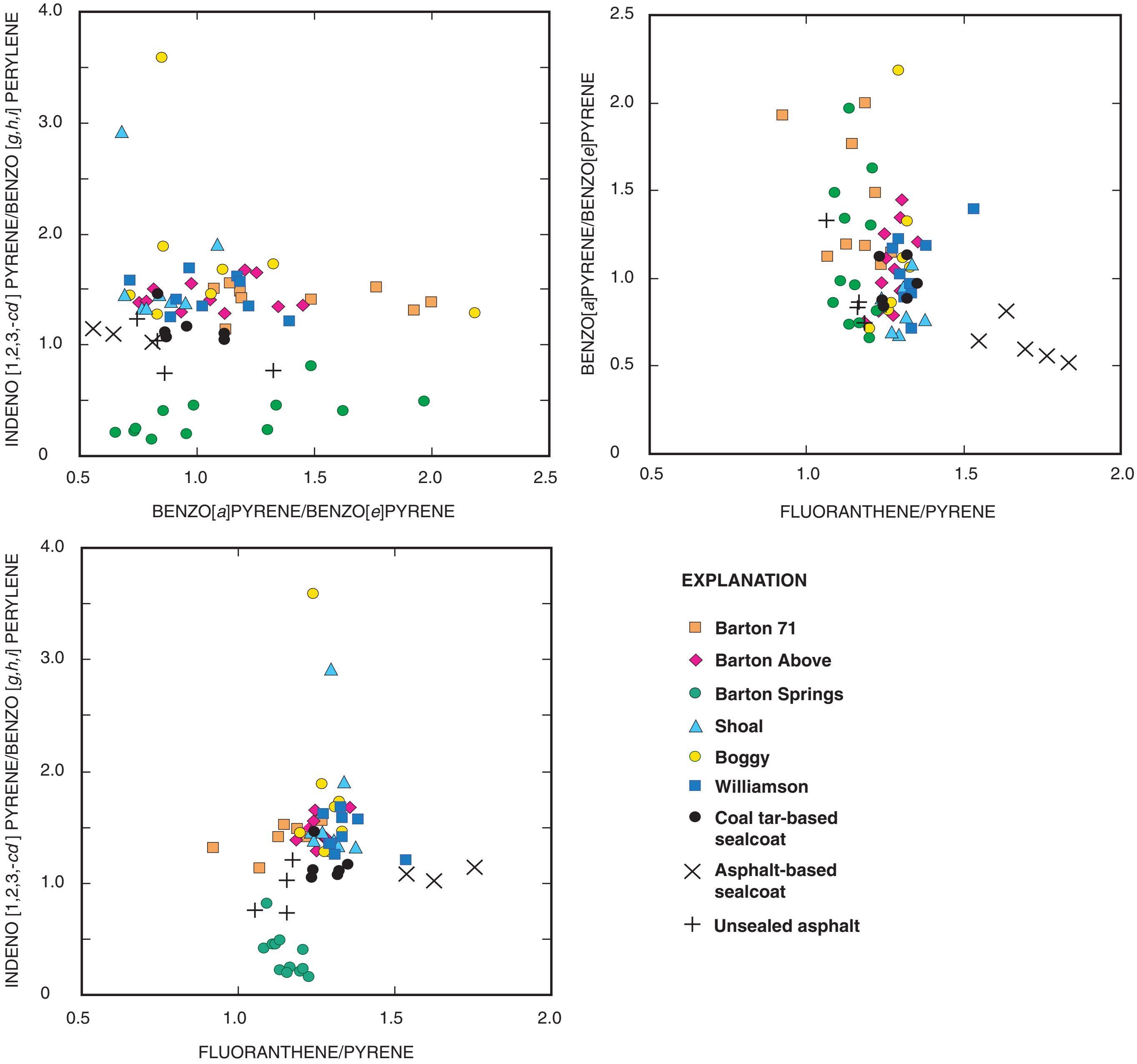


Figure 11. Comparison of polycyclic aromatic hydrocarbon (PAH) source ratios indicative of parking lot sealcoat and ratios in suspended-sediment samples in urban creeks, Austin, Texas, 1999–2004.

sediment because all soil and sediment contain some metals with a nonanthropogenic source—a baseline value—over which any anthropogenic signature is imprinted. Therefore even a very large variation in the anthropogenic contribution might cause a relatively small variation in the total concentration if the anthropogenic contribution is small in proportion to the baseline concentration. PAHs also occur naturally in the environment, but the baseline concentration usually is very low relative to anthropogenic concentrations. For example, the median of decadal-mean concentrations (1990s) of benzo[*a*]pyrene in sediment cores from seven undeveloped reference lakes across the United States was 7 µg/kg (Van Metre and Mahler, 2005). The benzo[*a*]pyrene concentrations reported here range from 2.9 µg/kg at Barton 71 to 14,000 µg/kg at Williamson. The organochlorine compounds have a baseline of zero, as they all are produced synthetically, which means that any measurable concentration has an anthropogenic source.

The amount of variability in the concentrations of the different analytes has important implications for prediction of event loads, because the computation of loads involves the concentration of both the contaminant and the suspended sediment. Of all the analytes of interest, SSC is the most easily and inexpensively analyzed. If the variation in the contaminant of interest is much less than the variation in the SSC, as it is for the metals, then the event load of the constituent will be largely proportional to the load of suspended sediment, with the constant of proportionality the average concentration of the metal. In that case, once the characteristic metal concentrations have been determined at a site, the load of each metal can be estimated on the basis of the suspended-sediment load. However, the characteristic metal concentration at a site might not be constant temporally—it might change as a result of changes in land use or traffic patterns, for example. Therefore actual metals concentrations should be determined periodically.

Trends in Concentration

To determine whether concentrations of contaminants in suspended sediment increased or decreased at the sites over the period of sampling, the nonparametric Kendall's tau test (Helsel and Hirsch, 1995) was used to indicate whether there was a statistically significant relation (p-value < .05) between concentration and time (that is the time series of dates of sample collection). Positive and negative taus signify upward and downward trends, respectively. The tau value is an indication of the amount of noise in the trend: a purely monotonic trend has a tau value of 1.0 (or –1.0), and trends with one or more values deviating from a monotonic trend will be between –1.0 and 1.0. Any trend with a p-value < .05 was assumed to be significant, regardless of the magnitude of the tau. For the purposes of rank-based trend testing, nondetections were assigned a zero concentration, giving them a lower rank than detected concentrations. If more than 60 percent of results for a constituent at a given site were nondetections, trend was not tested. For metals, all samples from each site were tested together, as there was no change in sampling protocol over the duration of the study. For organochlorine compounds and PAHs, samples processed using GFF were tested separately from those processed using PTFE filters. Series of less than four data points were not tested.

During 1999–2004, concentrations of metals associated with suspended sediment changed negligibly at some sites and decreased at others (table 3). At Barton 71, Barton Springs, Shoal, and Boggy there was no significant trend, either upward or downward, in concentrations of any of the eight anthropogenic metals. At Williamson there were downward trends in Cd and Ni, and at Barton Above there were downward trends in As, Cd, Cr, Pb, and Zn (fig. 12A; table 3). When these metals are normalized to Fe concentration, six of the seven trends, except As at Barton Above, are still significant, indicating a decrease in the anthropogenic contribution of these metals at these sites. Concentrations of Cd, Cr, Pb, and Zn associated with the most recently collected samples of suspended sediment at Barton Above are in the range of concentrations at Barton 71 (illustrated for selected metals in fig. 12B). It is not known what kind of activities might have led to these decreases, but a source or sources of metals might have been eliminated since the late 1990s, possibly as a result of some kind of infrastructure improvements. At those sites where metals are not trending, the relative lack of variability in metals concentrations indicates that SSCs could be used to estimate metals loads for events when metals chemistry is not measured.

Concentrations of organochlorines over the time period of the study for all but one compound at one site either had no significant trend (p-value > .05) or a downward trend (table 3). Only a small dataset could be trend tested because many organochlorine compounds had more than 60-percent nondetections at a given site (appendix 3.2). For the contaminants that could be trend tested, there were significant trends only at Barton Above and Shoal. Total DDT (the sum of DDT and its two breakdown products DDD and DDE) showed a downward trend at Barton Above for samples filtered using PTFE filters over the entire sampling period. Although this is not surprising, given that this compound was banned in 1972 and stocks in soils are therefore diminishing, no trends were seen at any of the other sites. Dieldrin increased in samples at Shoal filtered using GFFs over the sampling period at that site. This apparent trend, however, might not represent a true concentration increase, as the period of sampling was only 2.5 years, and use of this insecticide was voluntarily banned in 1987 (U.S. Environmental Protection Agency, 1990). Also of note is the lack of any trends detected in chlordane, although use of this insecticide was banned in 1988. Trends in organochlorine compounds might become clearer if analysis of samples processed with PTFE filters continues and the number of data points and the length of the record increases.

There were no statistically significant trends in PAHs at any site except Barton Above. At that site, concentrations of naphthalene in samples filtered using PTFE filters decreased over the corresponding 2.5-year period (November 2001 through February 2004; Kendall's tau of -.62, p-value = .05). None of the other PAHs at the site showed any temporal trend.

Loads and Yields

Loads (total mass of contaminant per rain event) and yields (total mass of contaminant per unit area of watershed per rain event) provide additional information about the contribution of contaminants by the different watersheds. Loads and yields for the events sampled are listed in appendixes 4 and 5. Load data are useful because they allow the total amount of a given constituent contributed by any one watershed to be compared to that contributed by another watershed. Although concentrations are affected by the dilution of contaminated sediment by clean sediment from, for example, bank erosion, addition of clean sediment does not affect the contaminant load, provided that the contaminant is not diluted to the point that the concentration is less than the LRL. Event loads were computed as the product of the flow-weighted composite concentration of the contaminant for each event and the load of suspended sediment for the event. The load of suspended sediment was

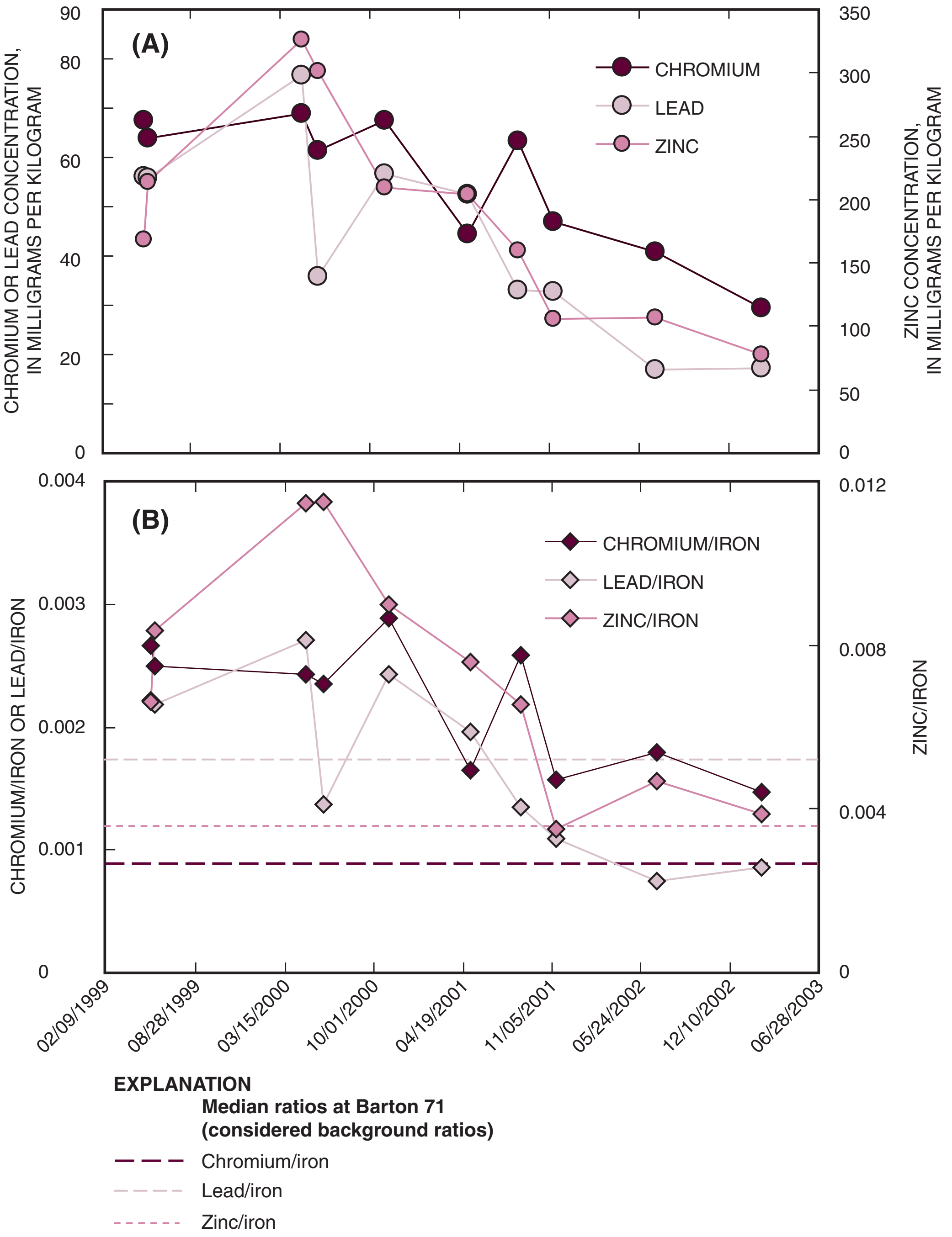


Figure 12. Downward trends in (A) concentrations of selected metals and (B) concentrations of selected metals normalized to iron at Barton Above, Austin, Texas, 1999–2003.

computed as the product of the flow-weighted composite SSC and the total event volume of flow. A stream can carry a small contaminant or sediment load and still have sediment with high concentrations of contaminants. If, for example, the SSC is low, or the volume of flow is small, the contaminant load might be low even if the suspended sediment is very contaminated.

Yield data are useful for comparison of land-use effects on contamination between watersheds. The yield expresses the mass of contaminant produced per unit area of watershed, and thus, unlike load, is independent of watershed size. For example, if there are two watersheds, one with a predominant land use producing very little contamination and the second producing much more contamination, they might still produce similar

Table 3. Temporal trends in metals and organochlorine compounds in suspended sediment in urban creeks, Austin, Texas, 1999–2004.

[Statistically significant trend (p-value < .05) in bold; positive and negative taus indicate upward and downward trends, respectively; PTFE, polytetrafluoroethylene filter; --, insufficient detections for testing; GFF, glass-fiber filter]

Sampling site	Metals								
	Iron	Arsenic	Cadmium	Chromium	Copper	Mercury	Nickel	Lead	Zinc
Barton 71									
Kendall's tau	-.42	-.02	-.36	-.20	-.31	-.22	-.38	-.20	-.24
p-value	.09	.93	.15	.42	.21	.45	.13	.42	.33
Barton Above									
Kendall's tau	-.24	**-.51**	**-.51**	**-.69**	-.20	-.24	-.42	**-.69**	**-.64**
p-value	.33	**.04**	**.04**	**.01**	.42	.36	.09	**.01**	**.01**
Barton Springs									
Kendall's tau	**-.44**	-.33	-.13	-.37	-.38	-.40	-.23	-.31	-.09
p-value	**.04**	.11	.54	.07	.07	.06	.27	.14	.68
Shoal									
Kendall's tau	-.33	-.50	.28	-.11	.17	.20	-.14	.17	.06
p-value	.21	.06	.30	.68	.53	.57	.60	.53	.83
Boggy									
Kendall's tau	**-.57**	.29	.07	.21	-.36	-.08	-.18	-.14	-.18
p-value	**.05**	.32	.80	.46	.22	.79	.53	.62	.53
Williamson									
Kendall's tau	-.11	-.50	**-.56**	-.28	-.20	-.15	**-.61**	-.44	-.06
p-value	.68	.06	**.04**	.30	.46	.67	**.02**	.10	.83

Site	Organochlorine compounds						
	Dieldrin	Chlordane	DDD	DDE	DDT	Total DDT	Total PCBs
Barton 71, PTFE							
Kendall's tau	--	--	--	-.59	-.59	--	--
p-value	--	--	--	.06	.06	--	--
Barton Above, GFF							
Kendall's tau	--	-.18	-.82	-.18	-.55	-.91	--
p-value	--	.71	.10	.71	.26	.06	--
Barton Above, PTFE							
Kendall's tau	--	--	--	**-.78**	**-.73**	**-.88**	--
p-value	--	--	--	**.01**	**.02**	**.01**	--
Shoal, GFF							
Kendall's tau	**.62**	.40	.11	.36	.50	.43	.40
p-value	**.03**	.17	.69	.22	.08	.14	.16
Boggy, GFF							
Kendall's tau	.14	.33	.41	.20	.60	.47	.60
p-value	.70	.35	.24	.57	.09	.19	.09
Williamson, GFF							
Kendall's tau	--	--	--	.60	--	.60	--
p-value	--	--	--	.14	--	.14	--
Williamson, PTFE							
Kendall's tau	--	--	--	-.91	--	-.55	--
p-value	--	--	--	.06	--	.26	--

loads of contaminants if the "clean" watershed was large and the "contaminated" watershed was small; however, when normalized to watershed area the large watershed would have a lower contaminant yield than the small watershed. For this study, event yield was computed as the event load divided by the watershed area.

Comparison of event concentrations, loads, and yields among sites indicates that high concentrations do not necessarily translate to high loads or yields (fig. 13), as noted above. The combination of higher SSC and a relatively smaller watershed for Boggy, in some cases, results in higher median event loads and yields than at the other sites. For example, although the highest metals concentrations were in either Shoal Creek or Williamson Creek, the highest median event loads and yields for metals were all in the Boggy Creek watershed. Similarly, the highest ΣPAH_{SQG} concentrations were in Williamson Creek, but the highest median event loads and yields for ΣPAH_{SQG} were in the Boggy Creek watershed. Median event loads and yields for chlordane also are highest in the Boggy Creek watershed, even though concentrations were higher in Shoal and Williamson Creeks (fig. 13).

Explanatory Variables

To better understand the occurrence of hydrophobic contaminants on suspended sediment, their relation to several possible explanatory variables was examined. The variables considered comprise: two measures of antecedent conditions (antecedent dryness period [days preceding last rainfall] and antecedent flow period [days preceding last streamflow]), which might affect the amount of contaminant buildup; rainfall depth and two measures of rainfall intensity (the event average and maximum), which might affect the efficiency with which contaminants are delivered to the creeks; SSC, which enters into the contaminant load computation and might affect the amount of dilution of contaminants; and land use, which might be related to source strength and thus affect concentrations and yields of contaminants.

Parametric rather than nonparametric tests were used because the interest was in the power of the explanatory variables for prediction of contaminant concentrations and event loads. For datasets with no censored data, linear correlations between the explanatory variables and the logarithms of measured concentrations and event loads were computed, and correlation was assumed to be significant at p-value < .05. The constituents included in this analysis were metals, some organochlorines at some sites (in particular DDE and total DDT), ΣPAH_{SQG}, and some individual PAHs at some sites. Datasets containing censored data, including multiply-censored data (that is, data censored at more than one reporting level), were analyzed using a parametric survival analysis method (StatSoft Inc., 2004), which efficiently uses the information available in multiply-censored data as part of the analysis (Helsel, 2005). Using the survival analysis method, linear regression was applied to the logarithms of the multiply-censored data; a statistically significant relation was identified as p-value < .05. The strength of the relation is reported as the likelihood coefficient r, analogous to Pearson's r in linear regression, and like Pearson's r varies from 0 (no correlation) to ±1 (perfect correlation). The likelihood coefficient is computed as the square root of the likelihood r^2, with algebraic sign identical to that for the slope of the relation (Helsel, 2005). The constituents included in this type of statistical analysis were organochlorines and individual PAH compounds at some sites; datasets were tested if they contained 50-percent censored data or less. The relation between DDD and total rainfall (fig. 14) is an example of the type of relation determined using this approach. Because of the relatively small number of events sampled, only one explanatory variable was included in each linear regression. In some cases, a high degree of correlation might be caused by an outlier that exerts a strong influence on the regression slope, as indicated by a high value for Cook's D (Helsel and Hirsch, 1995), where "high" is usually defined as a value exceeding $4/n$, where n is the number of cases. Datasets with significant correlations (p-value < .05) were tested for values with a high Cook's D, any outliers were removed from the analysis, and the dataset was retested. If p-value was no longer < .05, the relation was assumed not to be significant. Because Cook's D cannot be evaluated using the survival analysis method, the presence of outliers for datasets with multiply-censored data was determined subjectively by visual inspection.

Explanatory Variables and Concentrations

Significant relations between concentrations and total rainfall, average (rainfall) intensity, maximum (rainfall) intensity, antecedent dryness, and antecedent flow are listed in table 4. In addition, intercorrelations between individual metals and ΣPAH_{SQG} were tested (table 2).

Only two significant relations were seen for a concentration of any metal and total rainfall: a negative relation for Pb at Barton Above and a positive relation for As at Williamson. A negative relation with total rainfall suggests that the contaminant is at the surface and has an anthropogenic source, so that an increase in rainfall, which causes higher streamflow and more streambank erosion, results in a decrease in concentration. A positive relation with total rainfall suggests a soil or bed sediment source for the contaminant, so that an increase in rainfall results in increased erosion and mobilization.

Relations were seen between a metals concentration and average or maximum intensity of rainfall in five cases. In two cases the relation was negative, indicating a potential anthropogenic source for the metals, as high-intensity rainfall results in increased erosion of deeper soils. In three cases the relation was positive; the positive relation between rainfall intensity and Cu at Boggy or Cd or Pb at Williamson is difficult to explain.

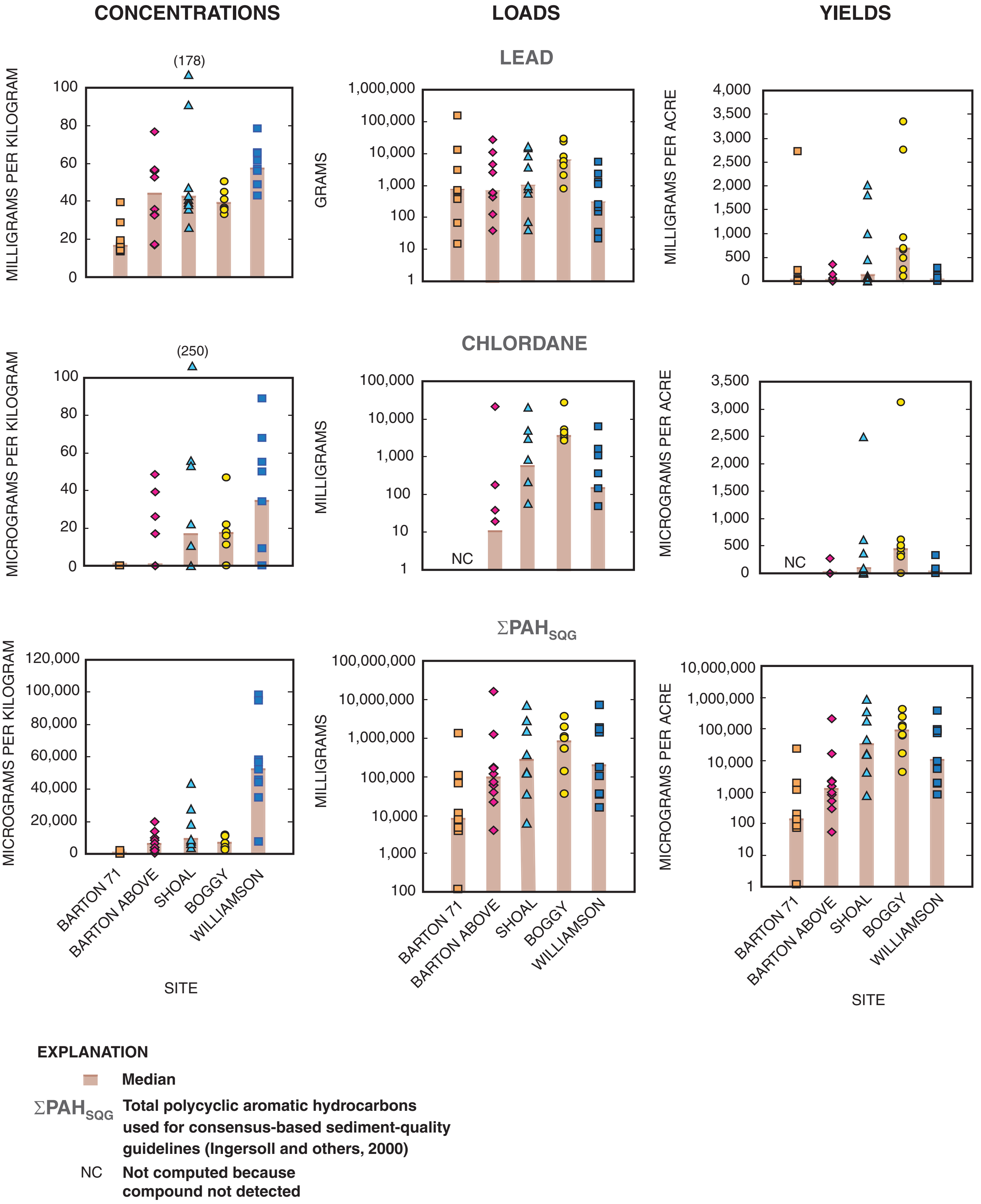


Figure 13. Comparison of concentrations, loads, and yields of selected contaminants in urban creeks, Austin, Texas, 1999–2004.

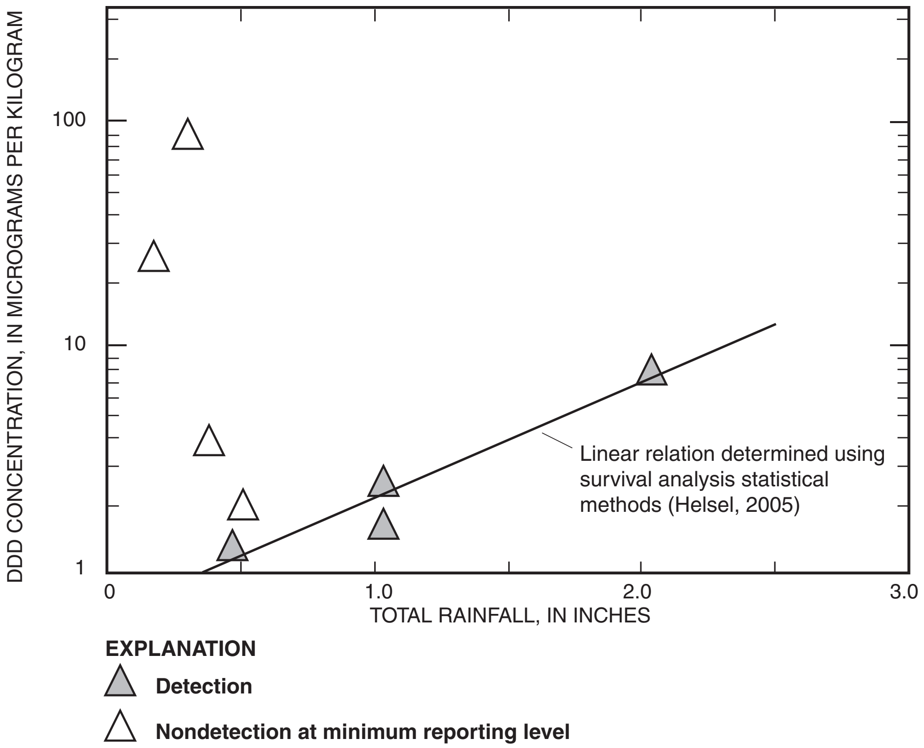


Figure 14. Relation between an explanatory variable and a dataset including multiply-censored data.

Overall, the most relations were seen for a metals concentration and one of the antecedent-condition variables that indicate length of time since rainfall or streamflow. This is consistent with contamination from metals with anthropogenic sources that build up on impervious surfaces, as noted by several researchers (Mason and others, 1999; Thomas and Greene, 1993; Yaziz and others, 1989). In particular, very strong positive relations were seen between antecedent dryness or flow and Cd and Cu (Shoal) and As and Cd (Barton Above). At Shoal, this is consistent with anthropogenic sources of Cd and Cu as indicated by the relation with Fe. The relations of As and Cd with Fe at Barton Above were less conclusive. The strong negative correlation between antecedent dryness and Pb at Boggy does not have an obvious explanation.

Correlation of organochlorines to the explanatory variables yielded few significant variable results. Rainfall was correlated with the logarithm of concentration only for DDD at Shoal ($r = .89$). In two cases, antecedent dryness and antecedent flow were correlated with logarithms of an organochlorine compound concentration: at Boggy for chlordane ($r = -.72$ and -.64, respectively) and at Shoal for DDD ($r = .63$ and .69, respectively). In two cases, antecedent dryness was correlated with the logarithm of an organochlorine compound concentration: at Barton 71 for DDE ($r = .64$) and at Boggy for dieldrin ($r = -.62$). Because of the small number of correlations and the lack of consistency in the sign of the correlation, no generalizations can be drawn about organochlorine concentrations and climatic variables.

Logarithms of SSC did not correlate with any of the predictive variables except at Barton Above. Logarithms of concentrations of ΣPAH_{SQG} did not correlate significantly with any of the predictive variables tested.

Relations Between Sediment Loads and Particle-Associated Contaminant Loads

To determine whether sediment load could be used to predict PAC loads for rain events, the relations between the two were investigated using either linear correlation for uncensored data or survival analysis for censored data. In all cases, the logarithms of sediment and contaminant loads were used.

All but one case for metals and ΣPAH_{SQG} event loads show statistically significant relations, most of which were extremely strong, with a median r^2 of .99 (range of .81 to 1.00) (fig. 15; table 5). In one case the statistical significance of the relation was the result of an outlier; when the outlier was removed the relation was no longer significant (p-value = .08). The linear relation is of the form

$$\begin{aligned} log\ (contaminent\ load) &= slope \times log\ (sediment\ load) \\ &\quad + intercept, \end{aligned} \tag{2}$$

and intercepts, slopes, and r^2 for each of the linear relations are shown in table 5. The As load was most strongly related to sediment load (median $r^2 = 1.00$), and ΣPAH_{SQG} was the least strongly related (median $r^2 = .87$) but still very strong.

Table 4. Correlation coefficients between logarithms of concentrations of suspended sediment, metals, and total polycyclic aromatic hydrocarbons used for consensus-based sediment-quality guidelines (ΣPAH_{SQG})[1] and explanatory variables for urban creeks, Austin, Texas, 1999–2004.

[Significant correlation (p-value < .05) in bold; correlation for which significance resulted from outlier shown in italics. SSC, suspended-sediment concentration; --, not applicable]

Sampling site and explanatory variable	log SSC	log Arsenic	log Cadmium	log Chromium	log Copper	log Mercury	log Nickel	log Lead	log Zinc	log ΣPAH_{SQG}
Barton 71										
Total rainfall (inches)	.47	.12	-.51	-.01	-.40	.20	.28	-.35	-.51	-.55
Average intensity (inches)	.46	**-.84**	-.25	-.60	-.58	-.06	.01	-.34	-.35	-.35
Maximum intensity (inches)	.59	-.60	-.58	-.48	**-.73**	-.05	.12	-.49	-.58	-.42
Antecedent dryness (days)	.17	-.21	-.02	-.06	-.20	-.03	-.04	-.10	-.14	-.53
Antecedent flow (days)	.21	.08	-.09	-.04	-.24	-.37	.06	-.21	-.30	-.65
Barton Above										
Total rainfall (inches)	.41	-.43	-.56	-.57	-.51	-.60	-.35	**-.76**	-.60	-.21
Average intensity (inches)	.34	-.23	-.36	-.16	-.25	-.21	-.47	.08	-.09	.16
Maximum intensity (inches)	.15	-.02	-.33	-.10	-.25	-.61	-.41	-.04	-.12	.14
Antecedent dryness (days)	**-.70**	.49	**.80**	.33	*.81*	.54	.51	**.68**	.52	.50
Antecedent flow (days)	-.35	**.79**	.43	.55	.24	-.10	.35	.39	.25	.48
Barton Springs										
Total rainfall (inches)	--	.03	.16	.13	-.17	-.57	-.01	.02	-.54	.27
Average intensity (inches)	--	-.51	-.05	-.50	-.60	.03	-.50	-.55	-.44	.18
Maximum intensity (inches)	--	.10	.22	-.01	-.18	.22	-.13	.02	-.32	.22
Antecedent dryness (days)	--	.32	.24	.23	.23	.38	.20	.36	-.04	.30
Antecedent flow (days)	--	.12	.17	.04	.38	.28	.25	.33	.33	.05
Shoal										
Total rainfall (inches)	.61	-.17	-.12	-.15	-.48	.67	-.02	-.36	-.49	-.06
Average intensity (inches)	.49	.22	-.12	-.44	-.34	.62	-.47	-.25	-.33	.04
Maximum intensity (inches)	.54	.32	-.11	-.33	-.39	.54	-.34	-.28	-.37	.04
Antecedent dryness (days)	-.63	.16	**.73**	-.06	**.70**	.14	.15	*.77*	*.73*	.29
Antecedent flow (days)	*-.73*	.08	*.81*	-.22	*.80*	.44	.05	*.86*	*.79*	.49
Boggy										
Total rainfall (inches)	-.02	-.64	.01	-.42	.17	-.19	.64	.11	-.03	-.15
Average intensity (inches)	.23	-.10	-.07	.39	**.73**	.46	.40	.11	.43	.26
Maximum intensity (inches)	.12	-.16	-.22	.04	.66	.46	.23	.15	.33	.09
Antecedent dryness (days)	.03	.40	.24	.50	-.01	-.05	.37	**-.75**	.00	-.18
Antecedent flow (days)	.21	.43	.18	.72	.12	.15	.36	-.66	.08	.00
Williamson										
Total rainfall (inches)	-.38	**.89**	.29	.59	.34	*.88*	*.71*	.04	.23	.06
Average intensity (inches)	-.33	.46	.56	.34	.53	.27	.44	**.76**	.59	-.26
Maximum intensity (inches)	-.43	*.71*	**.76**	.59	.71	.36	*.72*	*.76*	.70	-.15
Antecedent dryness (days)	-.03	-.45	-.22	-.08	.11	-.64	-.38	-.02	.29	-.47
Antecedent flow (days)	.30	-.49	-.05	-.33	.01	-.53	-.48	.16	.13	-.51

[1] From Ingersoll and others (2000).

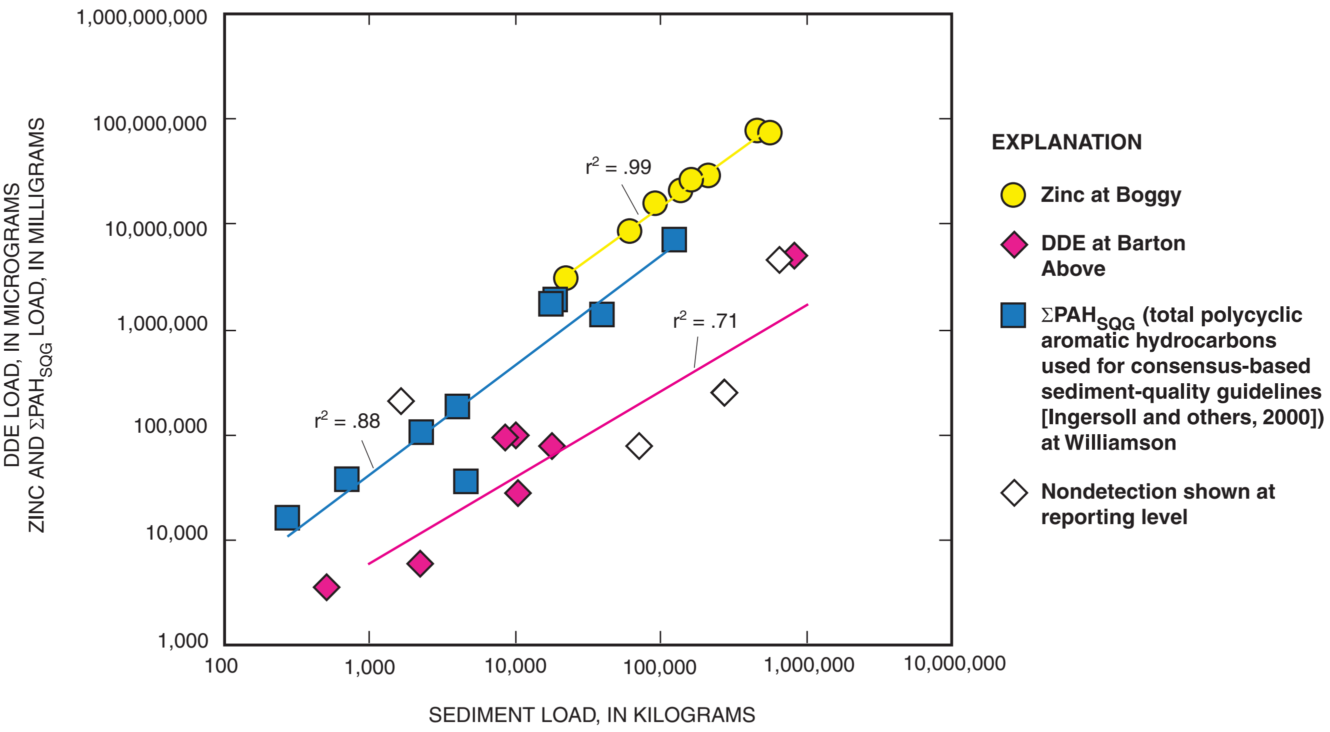


Figure 15. Representative examples of linear relations between sediment load and contaminant load in urban creeks, Austin, Texas, 1999–2004.

There were sufficient detections to investigate the relation between organochlorine load and sediment load in 17 cases, either using standard linear correlation or survival analysis. In each case, the logarithm of the contaminant load was tested against the logarithm of the sediment load. In 16 of the 17 cases, there was a statistically significant correlation (in the 17th case, the p-value was .057, so the relation could be considered significant at a 94-percent confidence level), and in all cases the relation was positive (fig. 15; table 5).

The relation between event loads of all three groups of contaminants and sediment load is not entirely unexpected, because contaminant loads are computed as the product of the contaminant concentration and the sediment load, which in turn is the product of the sediment concentration and the total volume of flow for the event. A statistically significant relation occurs because of much more between-event variability in the suspended-sediment load than in the contaminant concentrations. Thus once a characteristic contaminant concentration has been determined for a site, contaminant loads and yields for individual events can be estimated from measured sediment loads. Measurement of suspended-sediment load is relatively simple and inexpensive compared to measurement of contaminant loads; all that is required is analysis of a flow-weighted composite sample for SSC and stream discharge data for the event. Such prediction assumes, however, that the relations established through sampling remain relatively constant in time. If contaminant concentrations in the suspended sediment are changing over time, then the predictive relation to suspended-sediment load will change.

Relation Between Contaminant Yields and Land Use

Contaminant yields—the mass of contaminant contributed per unit area of watershed—are the most appropriate measure for comparing how the land-use characteristics of different watersheds affect the amount of contaminant produced. Ideally, mean annual particle-associated yields of contaminants from each watershed should be used in these comparisons. Unfortunately, annual yields are not available for the Austin creeks because not every event is sampled. Therefore, the possible relation between land use and yield for the different contaminant groups was explored through use of the event yields for each site. Because yield can vary greatly depending on the size of the event, the results might give only a general idea of possible relations and should be interpreted with caution.

Three measures of urban land use were compared to event yields of metals, organochlorine compounds (dieldrin, chlordane, and the DDTs), and ΣPAH_{SQG} at the five creek sites. The statistical parameter used was Spearman's rank correlation (Helsel and Hirsch, 1995), a nonparametric test that differs from computation of Pearson's test of linear correlation only in that the computations are done after the data have been converted to ranks. Spearman's rho is analogous to Pearson's r

Table 5. Parameters for linear correlations between logarithms of suspended-sediment load and loads of metals, organochlorine compounds, and total polycyclic aromatic hydrocarbons used for consensus-based sediment-quality guidelines (ΣPAH_{SQG})[1] for urban creeks, Austin, Texas, 1999–2004.

[Significant relation (p-value < .05) in bold; correlation for which significance resulted from outlier shown in italics; relation determined using multiply-censored data and survival analysis underlined. --, insufficient detections for testing]

Sampling site	Metals loads								
	Parameter	log Arsenic	log Cadmium	log Chromium	log Copper	log Mercury	log Nickel	log Lead	log Zinc
Barton 71	intercept	**1.05**	0.30	**1.73**	**1.43**	**-1.37**	**1.38**	**1.63**	**2.59**
	slope	**.97**	**.85**	**.97**	**.95**	**.99**	**.99**	**.92**	**.86**
	r^2	1.00	.97	.99	1.00	.99	.99	.99	.98
Barton Above	intercept	**1.40**	**.82**	**2.11**	**1.56**	**-.85**	**1.65**	**2.40**	**2.91**
	slope	**.92**	**.76**	**.91**	**.95**	**.91**	**.95**	**.82**	**.85**
	r^2	.99	.94	.99	1.00	.96	.99	.98	.98
Shoal	intercept	**1.08**	.62	**1.82**	**2.15**	**-1.60**	**1.42**	**2.49**	**2.95**
	slope	**.99**	**.82**	**1.01**	**.85**	**1.06**	**1.01**	**.82**	**.85**
	r^2	1.00	.93	1.00	.99	.99	1.00	.98	.99
Boggy	intercept	**1.33**	-.03	**1.86**	**1.32**	**-1.65**	**1.17**	**1.03**	**2.13**
	slope	**.94**	**.97**	**.98**	**1.01**	**1.07**	**1.05**	**1.11**	**1.01**
	r^2	.99	.91	.99	.99	.98	.97	.99	.99
Williamson	intercept	**1.12**	.01	**1.95**	**1.65**	**-1.48**	**1.63**	**1.92**	**2.81**
	slope	**.99**	**.97**	**.96**	**.95**	**1.09**	**.96**	**.96**	**.89**
	r^2	1.00	.96	1.00	.99	1.00	.98	.99	.99

Sampling site	Organochlorine compound and polycyclic aromatic hydrocarbon loads							
	Parameter	log Dieldrin	log Chlordane	log DDD	log DDE	log DDT	log Total DDT	log ΣPAH_{SQG}
Barton 71	intercept	--	--	--	**0.94**	--	--	**0.93**
	slope				**.79**			**1.10**
	r^2				.82			.87
Barton Above	intercept	--	--	--	**1.30**	--	**-1.62**	**1.02**
	slope				**.82**		**.84**	**.90**
	r^2				.71		.70	.81
Shoal	intercept	**0.90**	**2.97**	**-1.27**	**2.32**	**3.26**	**2.84**	*1.16*
	slope	**.89**	**.67**	**1.31**	**.71**	.47	**.69**	*.97*
	r^2	.91	.75	.87	.87	.40	.80	*.93*
Boggy	intercept	--	**-21.7**	-8.9	-2.61	**-1.79**	-.85	**.66**
	slope		**5.2**	2.8	**1.75**	**1.58**	**1.40**	**.93**
	r^2		.64	.63	.90	.86	.86	.86
Williamson	intercept	--	**-1.70**	--	**.83**	--	**1.15**	**.94**
	slope		**1.69**		**.93**		**.86**	**.84**
	r^2		.64		.81		.81	.88

---

[1] From Ingersoll and others (2000).

and is a measure of the linear correlation of the ranked data. The land-use types tested were total developed (equal to watershed area minus agriculture, undeveloped land, and park land), industrial, and residential (single family plus multifamily) (table 1) (Robert Botto, City of Austin, written commun., 2003). Developed nonresidential, transportation (streets and roads), and office+commercial+civic also were considered, but as Spearman's test is based on ranks and the watersheds are ranked identically for these three land-use types as they are for total developed, the correlation results were identical to those for total developed and are not shown.

All of the metals yields were significantly correlated (p-value < .05) with total developed and industrial, and in two cases with residential as well (table 6). The strongest correlations were with industrial land use, and the weakest were with residential land use. The relations might explain why, overall, metals contamination in the suspended sediment was relatively low, as the watersheds tested have a small amount of industrial land use (less than 10 percent) and a large amount of residential land use (24 to 43 percent). Hg correlated equally well with all land uses except residential, for which it had a somewhat lower value of rho.

Table 6. Results of Spearman's rank correlation test between event yields of contaminants and land use for watersheds of urban creeks, Austin, Texas, 1999–2004.

[Spearman's rho shown; significant relation (p-value < .05) in bold; strongest relation for each contaminant underlined; ΣPAH_{SQG}, total polycyclic aromatic hydrocarbons used for consensus-based sediment-quality guidelines (Ingersoll and others, 2000)]

Contaminant	Land use		
	Total developed	Industrial	Residential
	Metals		
Arsenic	**.42**	**.48**	.29
Cadmium	**.38**	**.44**	.26
Chromium	**.37**	**.44**	.25
Copper	**.39**	**.45**	.27
Mercury	**.42**	**.42**	**.36**
Nickel	**.33**	**.41**	.21
Lead	**.43**	**.48**	**.30**
Zinc	**.41**	**.46**	.28
	Organochlorine compounds		
Dieldrin	**.56**	**.45**	**.46**
Chlordane	**.62**	**.55**	**.45**
DDD	**.42**	**.51**	.20
DDE	**.66**	**.67**	**.44**
DDT	**.54**	**.70**	.24
Total DDT	**.63**	**.66**	**.42**
	Polycyclic aromatic hydrocarbons		
ΣPAH_{SQG}	**.67**	**.57**	**.51**

Of the organochlorine compounds, yields of dieldrin, chlordane, DDD, DDE, DDT, and total DDT were compared to land use; detections of PCBs were insufficient for testing. Nondetections were given an equal rank, lower than that of all detections. All of the compounds were significantly correlated with either total developed or industrial, but not residential (table 6). Dieldrin and chlordane were correlated with all three land uses but most strongly correlated with total developed; whereas the DDT compounds were more strongly correlated with industrial land use.

Yields of ΣPAH_{SQG} were significantly correlated with the three urban land-use types tested, with the strongest correlation for total developed (table 6). The strength of these correlations was greater than that for any of the metals. These results are consistent with other studies linking PAH concentrations and urban land use (Brenner and others, 2002; Hoffman and others, 1984; Oros and Ross, 2004; Van Metre and Mahler, 2005; Van Metre and others, 2003; Walker and others, 1999).

The variation and trend in contaminant event yields with increasing percentage of total developed land in the watersheds of urban creeks is shown in figure 16. A positive trend of higher yields with larger percentages of developed land is evident for metals, organochlorine compounds, and ΣPAH_{SQG}.

Evaluation of Relative Contamination

To compare the degree of overall sediment contamination at one site to that at another, a measure developed by MacDonald and others (2000) was used. This measure, referred to here as the contamination index (C_I), is based on the three different groups of contaminants with consensus-based SQGs (metals, organochlorine compounds, and PAHs). The C_I is the mean of the PEC quotients for each of the three contaminant groups. The PEC quotient is computed by first dividing the concentration of each element or compound that was measured in the sample (and deemed reliable by MacDonald and others [2000]) by its PEC. MacDonald and others (2000) did not include chlordane in their C_I because its predictive ability (73-percent correct for the PEC) was below their threshold of 75 percent. However, because chlordane detected in this study frequently exceeded the PEC, and because its predictive ability as determined by MacDonald and others (2000) was very close to their threshold of 75 percent, chlordane was factored into the C_I for this study. To obtain the mean PEC quotient for each contaminant group, the quotients for each contaminant group are summed and divided by the number of quotients; the mean of the three mean PEC quotients is the C_I. Constituents not analyzed were not included in the mean PEC quotient computations. Nondetections were treated as zero values. MacDonald and others (2000) determined that a C_I greater than 0.5 for bed sediment is a reasonable indicator of whether the sediment is toxic to benthic biota and were able to correctly predict toxicity in 85 percent of samples tested. Samples with a C_I of 1.0 or greater were toxic to one or more species of aquatic

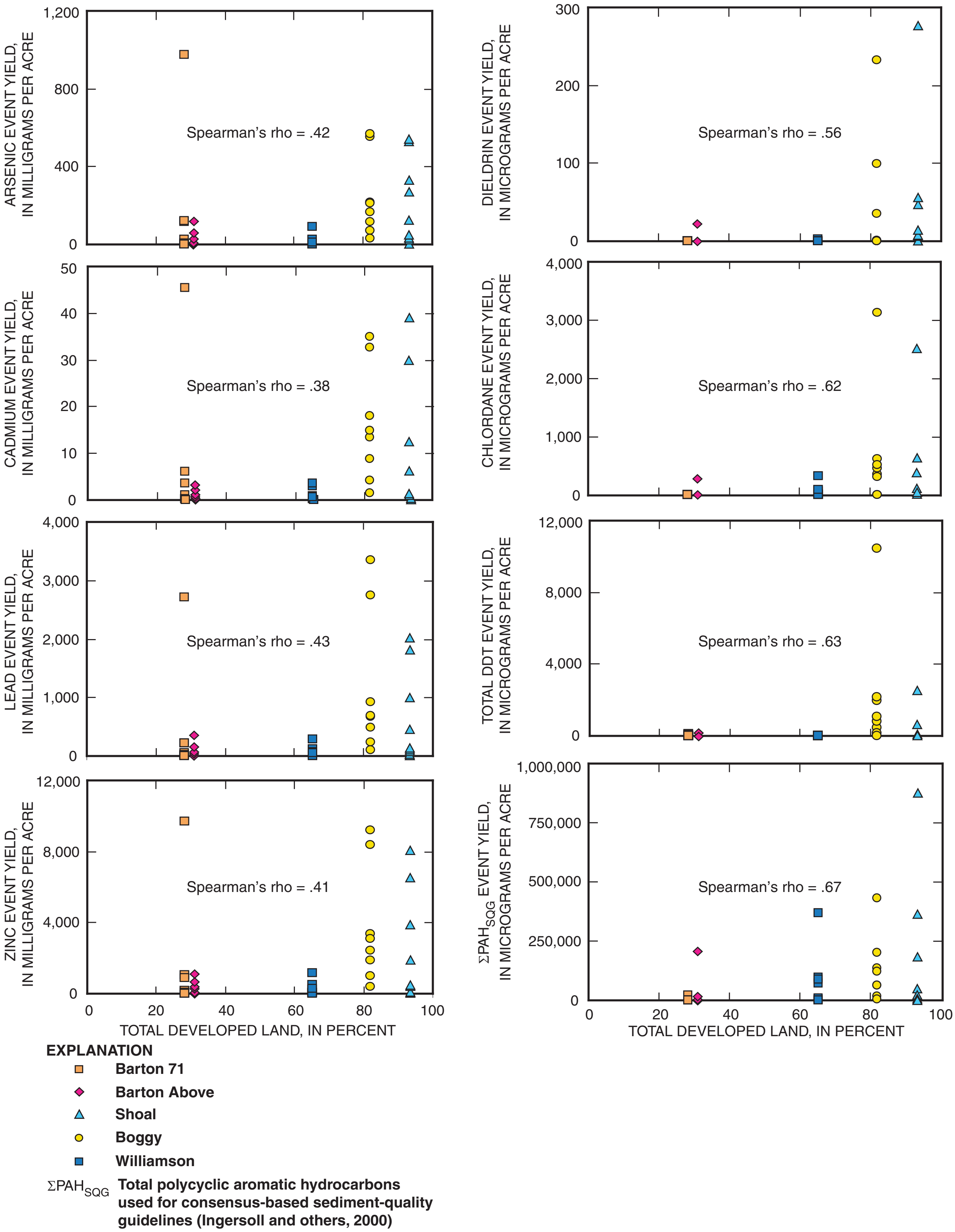


Figure 16. Representative relations between contaminant event yields and total developed land in watersheds of urban creeks, Austin, Texas, 1999–2004.

organisms in 92 percent of samples tested (MacDonald and others, 2000). One caution is that the SQGs were developed for bed sediment and not for suspended sediment in the water column, thus they should not be interpreted as a direct indicator of potential toxicity of the suspended-sediment samples collected here. The guidelines, however, provide a useful benchmark for indicating the degree of contamination at a site and for comparing overall suspended-sediment contamination among sites.

Of the 58 samples for which a C_I was computed, 17 have a C_I greater than 0.5 (fig. 17). These 17 samples are from the four urban sites: Barton Above, Shoal, Boggy, and Williamson. The median C_I from Williamson exceeds 0.5; the median C_I from Shoal is 0.47. The two sites with the lowest C_I are Barton 71 (median of 0.07) and Barton Springs (median of 0.12).

The three contaminant groups contribute different amounts to the C_I at the different sites (fig. 17). The metals contribute a more consistent amount to the C_I than the two other contaminant groups at all sites, likely reflecting the background metals concentration associated with the mineralogy of the sediment. PAHs contribute the most to the C_I overall and vary much more among sites. PAHs contribute almost nothing to the C_I at Barton 71 and Barton Springs but make up most of the contamination associated with suspended sediment from Williamson. Organochlorine compounds make almost no contribution to the C_I at Barton 71 and Barton Springs but, for some events at Shoal and Boggy, contribute more than the other two groups of contaminants.

Contaminant Concentrations in Context of Local and Nationwide Data

Contaminant concentrations on suspended sediment were compared to concentrations in bed-sediment samples collected from Town Lake to put them in a local context and were compared to sediment from the tops of cores from reservoirs around the country to put them in a nationwide context. Bed-sediment samples from Town Lake were collected by the USGS every year, beginning in 1991, from six sites (from upstream to downstream): at the Mopac bridge, at the Lamar Boulevard bridge, at the mouth of Shoal Creek, at the First Street bridge, at the I–35 bridge, and upstream from Longhorn Dam (fig. 18). Sampling was discontinued after 1994 at the Mopac bridge and mouth of Shoal Creek and discontinued after 2001 at the four other sites. Samples were collected from a seventh site in Town Lake by the City of Austin 3 times during 2003–04 and analyzed for PAHs; the median of those data are included in the comparison here. Bed sediments from other lakes and reservoirs across the United States have been collected as sediment cores since 1992 as a part of the USGS National Water Quality Assessment (NAWQA) Program (Van Metre and Mahler, 2004). Samples from the upper parts of these cores, representing sediment deposited from the 1990s to the present (2001), from 35 (metals) and 38 (organochlorines and PAHs) of these water bodies enable evaluation of the Austin suspended sediment in a nationwide context (Mahler and others, 2006; Van Metre and Mahler, 2005).

Concentration ranges and medians of selected contaminants in suspended-sediment samples from the five creek sites and Barton Springs and in bed-sediment samples from Town Lake are shown in table 7. Suspended-sediment data from all sites where the contaminant was detected in at least one sample were included in the analysis. In determining the median concentration, which is based on ranks, nondetections are ranked as less than any detection (Helsel and Hirsch, 1995). Results from all of the suspended-sediment (LVSS data) surface-water sites are included here, not just those from streams that contribute to Town Lake.

The median Cr concentration at all of the suspended-sediment sites exceeds that in Town Lake bed sediment by a factor of 5 or more; the median Zn concentration at five of the six sites exceeds that in Town Lake bed sediment as well (table 7). The median Pb concentration at all of the suspended-sediment sites, however, is less than that for Town Lake bed sediment.

DDT was banned in 1972, but nonetheless was detected in several suspended-sediment samples and in bed sediment from Town Lake. The highest parent DDT concentration detected in any of the suspended- or bed-sediment samples was in a suspended-sediment sample from Shoal Creek (71 μg/kg), and the median value at two of the sites with urbanized watersheds is higher than that for Town Lake bed sediment. In contrast to parent DDT, median concentrations of total DDT in the suspended sediment at all sites except one are less than those in the Town Lake bed sediment. The difference between concentrations of parent DDT and total DDT in suspended sediment and bed sediment suggests that less degradation of DDT has occurred in the suspended sediment (which represents recently mobilized soils) than in the bed sediment. Parent DDT degrades primarily into two products, DDE and DDD. By computing the ratio of DDT to total DDT, the amount of degradation that the parent compound has undergone can be compared among sites. The median DDT/total DDT ratio at two of the three creek sites where DDT is detected regularly is 0.39 and 0.45 (table 7); in other words, about 40 percent of the total DDT is still in the form of the parent compound. In contrast, the median ratio in the Town Lake bed sediment was 0.10, which means that only about one-tenth of the total DDT is in the form of the parent compound. A high DDT/total DDT ratio is not necessarily an indication of recent DDT use, but instead probably is evidence of persistence in soils and recent mobilization, as DDT is thought to break down more rapidly in aqueous environments than in soils (Nowell and others, 1999). Similar phenomena have been observed in other urban watersheds in Texas and New England (Van Metre and Mahler, 2004).

Although historically high concentrations of DDT in Town Lake have been ascribed to a large spill of DDT and other pesticides into Town Lake in the early 1960s (Carson, 1962; Van Metre and Mahler, 1999), the median total DDT

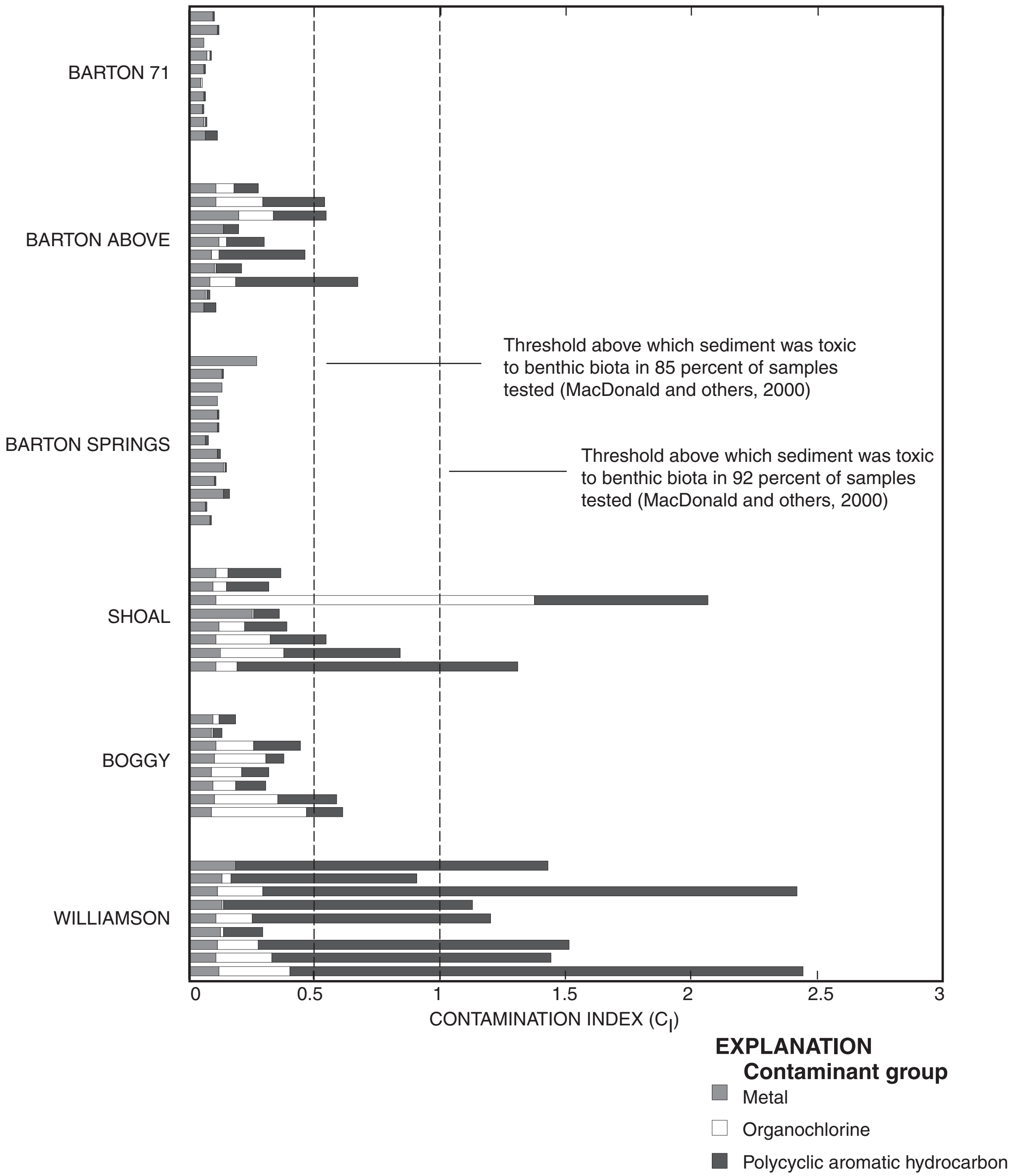


Figure 17. Comparison of sediment contamination index (C_I) and proportion of C_I contributed by each contaminant group in urban creeks, Austin, Texas, 1999–2004.

concentrations at Shoal and Boggy of 26 and 49 µg/kg, respectively, are similar to the median concentration of 44 µg/kg in the Town Lake bed sediment. The suspended-sediment data demonstrate that Austin watersheds are continuing to deliver DDT to the Colorado River above and below Longhorn Dam.

Chlordane, banned in 1988, was detected at four of the creek sites and in Town Lake bed sediment (table 7). The continued loading of chlordane to Town Lake is of concern in light of the history of chlordane-contaminated fish in the lake. A fish-consumption advisory for Town Lake, issued in 1990 by the Texas Department of Health on the basis of elevated chlordane concentrations in fish, was lifted in 1999 because “the latest tests found no chlordane or insignificant levels of chlordane in fish samples from the lake” (Texas Department of Health, 1999). The continuing presence of chlordane at elevated concentrations in both bed sediment and influent suspended sediment, however, indicates that fish continue to be exposed to chlordane.

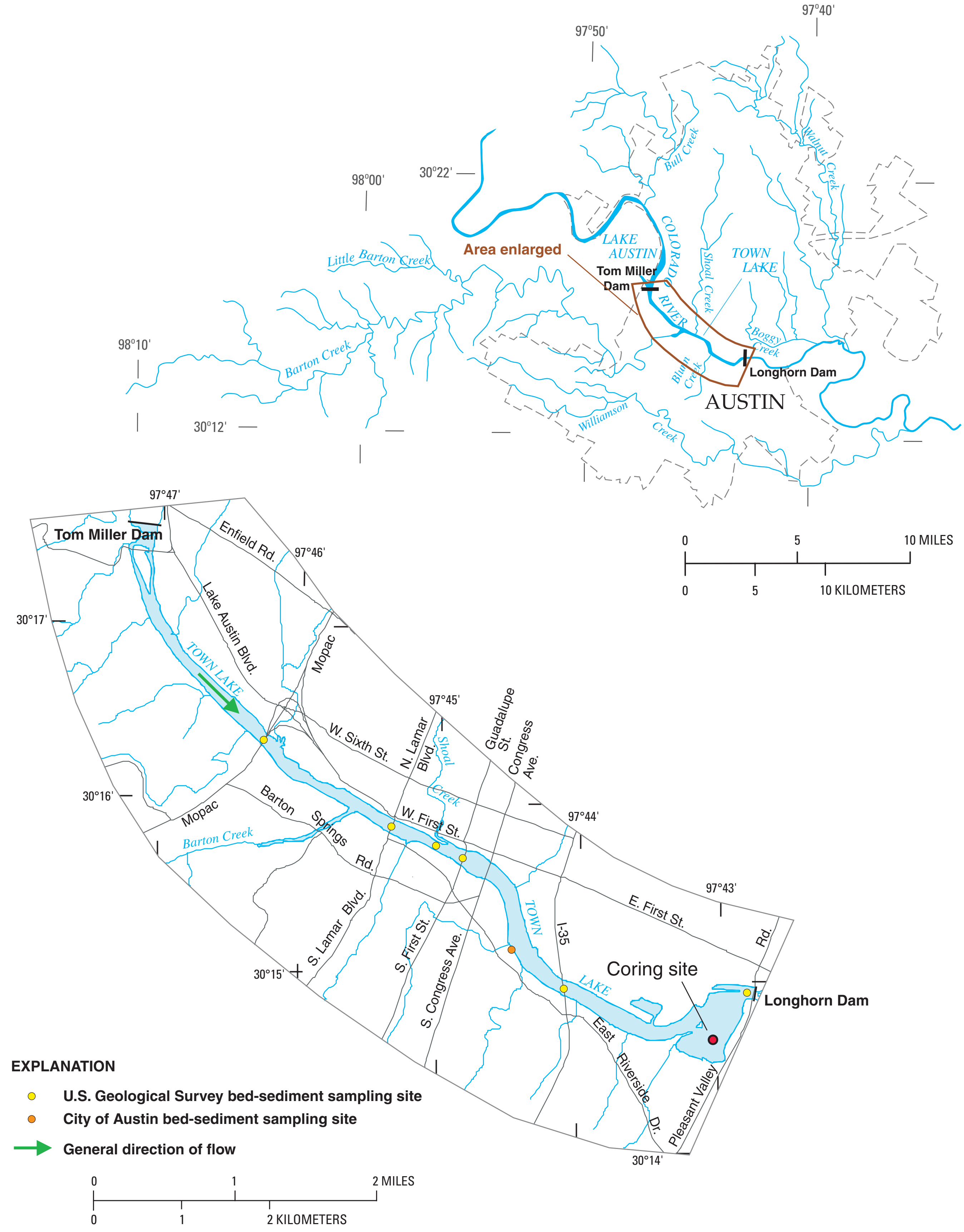


Figure 18. Bed-sediment sampling sites in Town Lake, Austin, Texas.

Table 7. Ranges and medians of contaminant concentrations in large-volume suspended-sediment samples from urban creeks (1999–2004) and in bed-sediment samples from Town Lake (1991–2004), Austin, Texas.

[mg/kg, milligrams per kilogram; µg/kg, micrograms per kilogram; ND, not detected; PAH, polycyclic aromatic hydrocarbons]

	Chromium (mg/kg)		Lead (mg/kg)		Zinc (mg/kg)		Chlordane (µg/kg)	
	Range	Median	Range	Median	Range	Median	Range	Median
Barton 71	28–67	36	13–39	16	55–271	66	ND	ND
Barton Above	30–69	62	17–77	44	77–326	186	ND–48	ND
Barton Springs	36–83	67	13–32	23	50–133	100	ND	ND
Shoal	61–94	70	25–196	42	105–607	183	ND–250	22
Boggy	51–69	53	33–50	39	126–172	144	ND–47	19
Williamson	57–88	65	43–79	57	175–466	237	ND–89	34
Town Lake bed sediments (n = 51)	1–21	7	10–170	65	20–520	80	ND–150	32

	DDT (µg/kg)		Total DDT (µg/kg)		DDT/ total DDT		Total PAHs (µg/kg)	
	Range	Median	Range	Median	Range	Median	Range	Median
Barton 71	ND–7.6	ND	ND–12	0.23	ND–0.9	ND	96–5,600	615
Barton Above	ND–13	ND	ND–25	7.1	ND–.69	ND	1,400–50,000	16,000
Barton Springs	ND	ND	ND	ND	ND	ND	86–2,100	450
Shoal	ND–71	7.6	5.4–300	26	ND–.69	0.45	9,800–110,000	21,000
Boggy	ND–65	17	2.4–160	49	ND–.50	.39	4,100–30,000	13,000
Williamson	ND–3.3	ND	ND–19	2.6	ND–1.00	ND	28,000–210,000	110,000
Town Lake bed sediments (n = 51)	.3–41	3.75	4.9–226	44	.01–.79	.10	[1]6,000–65,000	[1]10,000

[1] Five data points only.

PAH concentrations were measured in Town Lake bed sediment from four sites one time (2001) by the USGS. In addition, the City of Austin analyzed PAHs in bed sediment from a fifth site 3 times during 2003–04; these samples were analyzed by U.S. Environmental Protection Agency (1996) method SW 8270C. Method SW 8270C quantifies only a subset of the PAH compounds analyzed for in USGS samples. The relation between total PAH in USGS bed-sediment samples using the USGS analytical methods and the sum of the SW 8270C PAH compounds was determined by linear regression ($r^2 > .99$). This relation then was used to estimate total PAH concentrations for the City of Austin bed-sediment samples. The median concentrations of total PAH in suspended sediment at three of the sites with urbanized watersheds (Barton Above, Shoal, and Boggy) are of a similar magnitude to that in Town Lake (table 7). The samples from the five sites in Town Lake have a spatial pattern, increasing 12-fold from the Lamar Boulevard bridge to the First Street bridge, and then generally decreasing downstream to Longhorn Dam. The median concentrations at Barton Above and Shoal are only about one-quarter to one-third the Town Lake bed-sediment concentration reported for the First Street bridge sampling site of 65,000 µg/kg, which suggests that there is an important source of PAHs to Town Lake other than Barton Creek or Shoal Creek between Lamar Boulevard and First Street. Other factors also could contribute to the variation in concentrations, for example, variable patterns of deposition of sediment entering from the creeks or variations in organic carbon content. The median total PAH concentration in suspended sediment from Williamson Creek exceeds the median concentration in Town Lake by a factor of about 10.

On the basis of presence or absence, the LVSS data compare well with the Town Lake data. The legacy pollutants occurring in Town Lake (chlordane, DDT) also occur in suspended sediment in the two influent creeks (Barton and Shoal). The occasional flushing of sediment from Town Lake by floods coming down the Colorado River does not eliminate the contaminated bed sediment in the lake, as new contaminated sediment is delivered to Town Lake each time the urban creeks flow. Median concentrations of organochlorine compounds (chlordane and total DDT) in Town Lake bed sediment are higher than those in suspended sediment at Barton Above and Shoal. This indicates that other urban creeks in Austin that feed into Town Lake but were not sampled for this study might be transporting sediment with relatively high concentrations of organochlorine compounds to Town Lake. To make a quantitative comparison of occurrence in the creeks and the lake would require data on incoming loads from these other creeks, particularly those with large watersheds, and data sufficient to estimate annual loads rather than just event loads.

To put concentrations of hydrophobic contaminants associated with suspended sediment in the Austin creeks into a broader context, they are compared to concentrations in sediment from 38 lakes and reservoirs across the United States sampled by the USGS NAWQA Program (Van Metre and

Mahler, 2005). These lakes and reservoirs represent a range of land uses from remote (no human development) to 96-percent urban land use. Sediment cores collected from each lake were sectioned into discrete intervals and analyzed for metals, organochlorine compounds, PAHs, and date markers such as cesium-137. Dates were estimated for each sediment interval (Van Metre and others, 2004). The data presented here represent the mean concentration of each contaminant for those sediment intervals deposited in 1990 or later (Mahler and others, 2006; Van Metre and Mahler, 2005).

Concentrations of contaminants in suspended sediment can be compared to those in lake and reservoir bottom sediment. Van Metre and Mahler (2004) compared suspended sediment from five urban streams to tops of cores from reservoirs immediately downstream. They found that metals concentrations generally were similar between the two media but organic contaminant concentrations in suspended sediment were similar to or higher than those in the cores. Additionally, Van Metre and others (2003) analyzed the same contaminants analyzed for this study in suspended sediment (11 sites) and bed-sediment cores (three sites) from Fort Worth, Tex., and found that contaminant concentrations in suspended- and bed-sediment samples fit a similar increasing relation with increasing urban land use.

In the lake and reservoir core sediment, concentrations of contaminants follow a log-linear relation with urban land use, to which concentrations in suspended sediment can be compared (fig. 19). For this analysis, nondetections were assigned a concentration of 1.0 so they could be plotted on a log-linear plot. Town Lake is included in the sediment-core dataset, with land use computed for its watershed downstream from Tom Miller Dam.

For the metals, the relation between Cr and land use is very weak but much stronger for Pb and Zn, two metals associated with urban contamination (Callender and Rice, 2000). In general, concentrations of Cr, Pb, and Zn in the Austin suspended sediment are less than typical nationally for the associated amount of urban land use. This might occur because Austin has a relatively low amount of industrial land use, which often is associated with elevated metals concentrations (Chalmers and others, 2007). Additionally, Austin is not near any mining or smelting facilities, which often are associated with metals contamination.

In the national sediment core data, chlordane and total PCBs, which have or had many urban sources and few rural sources, correlate well with urban land use; total DDT, which was used heavily in agricultural areas and even in some forested areas, has a weaker relation (fig. 19) (Van Metre and Mahler, 2005). For these three organochlorine compounds, concentrations in Town Lake are higher than would be predicted on the basis of land use in its watershed, but concentrations in suspended sediment are lower for all sites except Barton Above. At Barton Above, concentrations of total DDT and chlordane are elevated on the basis of land use in this watershed. The low concentrations of total PCBs at all sites probably are a result of the relatively low amount of industrial land use in the watersheds.

In the national sediment core data, concentrations of PAHs (ΣPAH_{SQG}) correlate well with urban land use, increasing by a factor of about 50 as urbanization increases from 0 to 100 percent (fig. 19). In a national context, ΣPAH_{SQG} concentrations in suspended sediment from Barton 71 and Boggy are lower than would be predicted on the basis of land use in their watersheds, and concentrations at Shoal are similar to those in similar land-use settings. Concentrations at Barton Above and Williamson, however, are about 10 times higher than would be predicted on the basis of land use in their watersheds. This indicates either that there is an anomalous number of sources of PAHs in these watersheds, that characteristics of the watersheds are such that sediment contaminated with PAHs is efficiently delivered to the creeks, that dilution by “clean” soil is less, or some combination of these factors.

The Barton Sites—Barton Creek and Barton Springs

Water quality in Barton Creek and Barton Springs is of particular interest for several reasons. The aquifer feeding Barton Springs is a federally-designated sole-source aquifer and provides drinking water for about 50,000 people (Hunt and others, 2004); Barton Creek and Barton Springs contribute to the City of Austin drinking-water supply obtained from Town Lake; Barton Springs pool is one of Austin’s major tourist attractions; Barton Springs is home to the federally-listed endangered Barton Springs salamander; and the watersheds of the creeks that recharge the Barton Springs segment of the Edwards aquifer, including Barton and Williamson Creeks, are undergoing rapid urbanization. This section focuses on differences in water quality in the upstream and downstream parts of Barton Creek and the potential effects of urbanization on Barton Springs and Barton Springs pool.

The Barton Springs segment of the Edwards aquifer primarily is recharged by infiltration through the beds of six major creeks: Barton, Williamson, Slaughter, Bear, Little Bear, and Onion (fig. 20). The hydrologic relation between the creeks and the aquifer is discussed in detail in Slade and others (1986). Water in the upper reaches of Barton Creek flows over the surface until it reaches the recharge zone, where it infiltrates and recharges the aquifer. Water infiltrating through the bed of Barton Creek in the upstream part of the recharge zone (above USGS streamflow-gaging station 08155300 Barton Creek at Loop 360, Austin; fig. 20) discharges from Cold Springs, whereas water infiltrating farther downstream discharges from Barton Springs (Hauwert and others, 2001). During periods of high ground-water levels, water in the most downstream part of the creek (in the vicinity of Barton Above) does not infiltrate because the water table is above the creek bed (Slade and others, 1986).

Land use within the Barton Creek watershed follows a gradient from largely ranch land and rural residential in the upper part of the watershed to intensive urban in the lower part of the

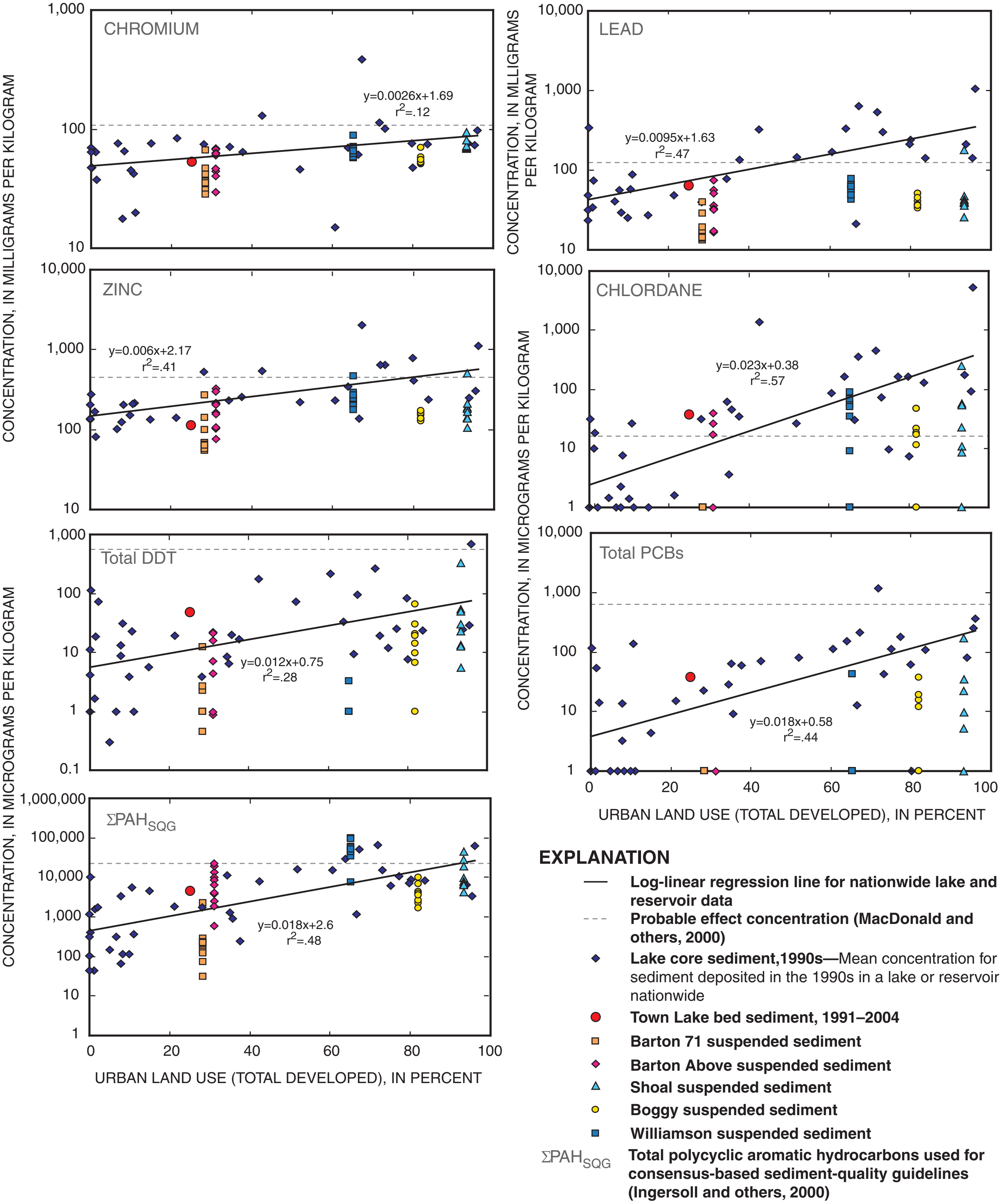


Figure 19. Comparison of suspended-sediment concentrations, Austin, Texas, 1999–2004, to national data for recently deposited (1990s) bed-sediment data in lake and reservoir sediment cores across an urban land-use gradient indicated by total developed land. Lake and reservoir data from Mahler and others (2006) and Van Metre and Mahler (2005).

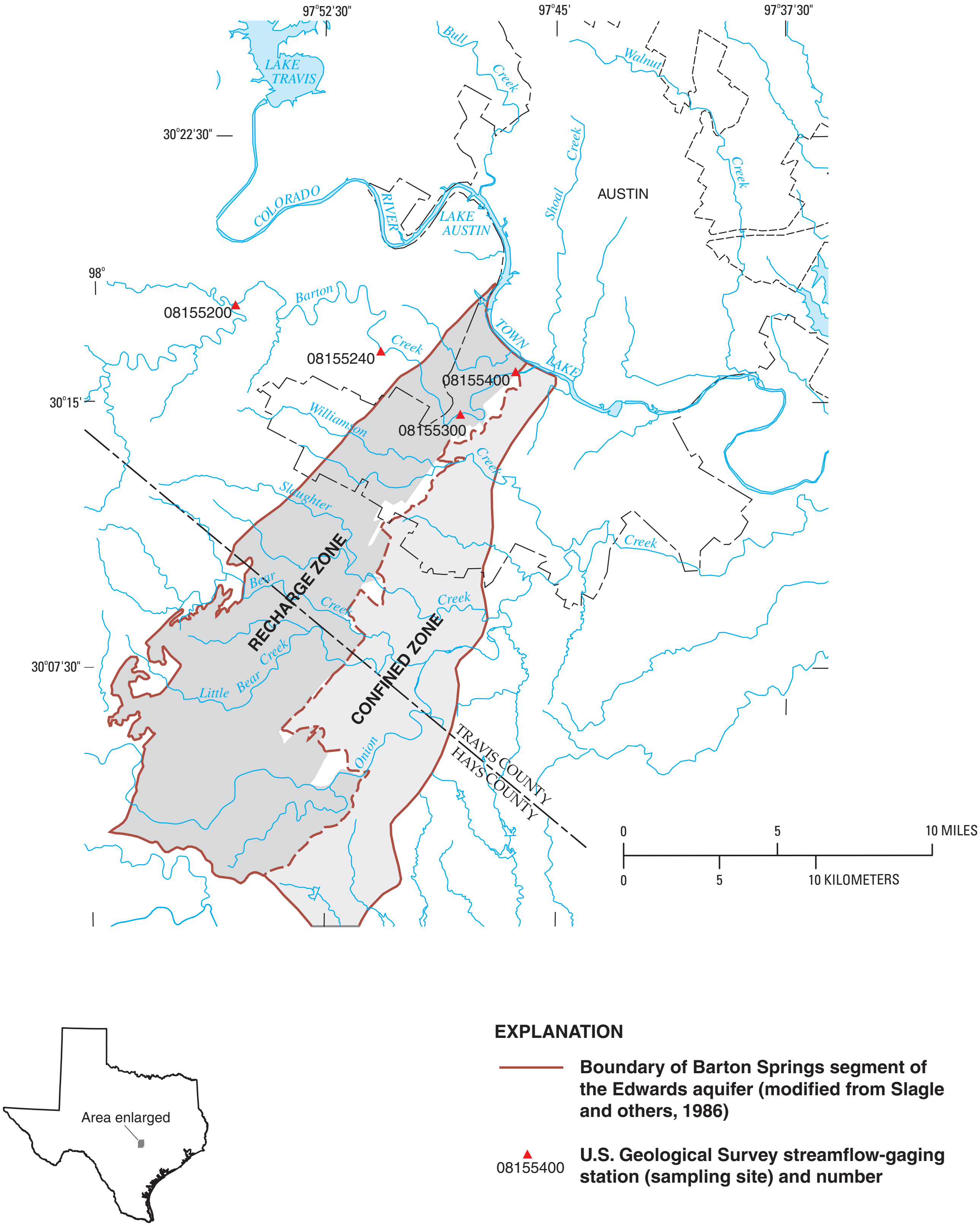


Figure 20. Location of Barton Springs segment of Edwards aquifer and major creeks in Austin, Texas, area.

watershed. Under some hydrologic conditions, for example when antecedent moisture conditions are dry and a rain event is minor or localized, the water and sediment chemistry at Barton Above (station 08155400, fig. 20) are dominated by runoff from the lower Barton Creek watershed. In contrast, when a major rain event occurs, water can flow from Barton 71 (station 08155200, fig. 20) all the way to Barton Above, and flow at Barton Above is assumed to integrate the effects of the entire watershed. The time required for the flood pulse to travel from Barton 71 to Barton Above ranges from 5 to 8 hours for a large rain event (flow of 2,000 ft^3/s or greater) to about 12 hours for a smaller rain event (flow of 500 ft^3/s or less) (Milton Sunvison, U.S. Geological Survey, unpub. data, 2003). Local runoff often will result in a peak in the hydrograph at Barton Above early in the event, followed by a larger peak several hours later that reflects flow originating farther up the watershed.

Contaminant concentrations and C_I increase from Barton 71 to Barton Above (figs. 6, 8, 9, 17; appendix 3). Although the upper part of the Barton Creek watershed still is lightly developed, the lower part of the watershed from the Loop 360 bridge (station 08155300, fig. 20) downstream has a larger amount of urbanization. For example, in 2001 (Robert Botto, City of Austin, written commun., 2003), the watershed upstream from Barton 71 was zero multifamily residential, 0.3-percent commercial, and 0.05-percent office, whereas the watershed between Barton Above and the Loop 360 bridge was 12-percent multifamily residential, 5-percent commercial, and 8-percent office. On the basis of analysis of flow data from the four streamflow-gaging stations on Barton Creek (fig. 20), of the 11 events sampled at Barton Above, five consisted entirely of local flow (May 17–18, 1999; May 25–26, 1999; May 6, 2001; August 26, 2001; and November 15, 2001); one consisted of runoff from the part of the watershed downstream from Lost Creek Boulevard (May 1–2, 2000, measured at USGS streamflow-gaging station 08155240 Barton Creek at Lost Creek Boulevard near Austin); one consisted of runoff downstream from State Highway 71 (June 9, 2000); three consisted of runoff from the entire watershed (November 3, 2000; June 30, 2002; and January 17, 2004); and one consisted of runoff upstream from State Highway 71 only (February 20, 2003).

If station 08155240 (fig. 20) is considered a boundary between the less urbanized area upstream and the more urbanized area downstream, the samples can be divided into two groups: those representing sediment coming from the urbanized part of the watershed (Group I, six samples), and those representing sediment with a contribution from the less urbanized part of the watershed (Group II, five samples). The median concentrations of all metals, except Cd and Cu, were higher for Group I, with the median Pb concentration 104-percent higher. (Median concentrations of Cd and Cu were about the same.) However, only the difference in Hg (0.09 mg/kg for Group I, 0.04 mg/kg for Group II) was statistically significant (t-test, $p\text{-value} < .05$), and might result from a geologic difference in the contributing areas.

Differences also were evident in detections of organochlorine compounds. Chlordane and DDD were detected in four and two, respectively, of the six Group I samples and not detected in any of the five Group II samples. DDE was detected in all of the Group I samples and in only two of the Group II samples; DDT was detected in four of the Group I samples and in only one of the Group II samples. No organochlorine compounds were detected in the single sample representing runoff downstream from State Highway 71, which indicates that the source of these legacy pollutants is in the part of the watershed downstream from Lost Creek Boulevard.

The median ΣPAH_{SQG} concentration for Group I was 9,400 μg/kg, about 3.7 times higher than the median concentration for Group II; however, the highest ΣPAH_{SQG} concentration detected at Barton Above (22,400 μg/kg) occurred in a sample representing runoff from the entire watershed (January 17, 2004). If this last high concentration is included in the dataset, the difference between the two groups is not statistically significant, but if the high concentration is dropped, the concentrations in Group I are significantly greater than those in Group II ($p\text{-value} = .03$). It is possible that event size might bias the results, as the median peak flow at Barton Above for Group II is about 3.8 times greater than that for Group I (although the difference between the two groups is not statistically significant). The larger events, however, do not appear to result in dilution of contaminants by clean sediment from streambank erosion, as the median SSC for Group I (urbanized) in fact is greater than that for Group II (less urbanized), although again there is no statistically significant difference between the two groups. These results indicate that the density of development on the part of the watershed contributing runoff to the creek might have a direct effect on the degree of contamination of the suspended sediment.

Sediment discharging from Barton Springs is much less contaminated than that in nearby urban creeks. A review of spring sediment chemistry during 2000–2002, including sediment discharging from the three smaller springs that make up the Barton Springs system (Upper Barton, Eliza, and Old Mill [fig. 1]), and a comparison to suspended sediment in urban streams, is presented in Mahler (2003). In general, concentrations of metals associated with spring sediment tend to be higher than those in the urban streams, and concentrations of organochlorine compounds and PAHs tend to be much lower or not detected. Median concentrations of As, Hg, Ni, and Zn in sediment discharging from Barton Springs were higher than those for the urban streams. Concentrations of Cu and Ni in sediment from Barton Springs exceeded the TEC more frequently than concentrations from any other site. In no case, however, did concentrations of a metal from Barton Springs exceed the PEC. Concentrations of all metals except Cd and Hg in sediment from Barton Springs are well correlated with Fe concentrations (fig. 7; table 2), with a correlation coefficient r from .74 (Cu) to .97 (As) that indicates the concentrations of the metals are related to the natural geochemistry of the sediment rather than to anthropogenic sources. In a few cases high outliers (for example, one for Cr, one for Cu) suggest isolated incidences of an anthropogenic source (fig. 7). Cd concentrations are low and relatively invariant, regardless of Fe concentration.

Hg concentrations vary greatly, are independent of Fe concentration, and are low overall, with a median of 0.07 µg/kg compared to a median of 0.16 µg/kg in sediment deposited since the 1990s in 35 lakes and reservoirs in the United States (Mahler and others, 2006).

The only organochlorine compound detected in sediment from Barton Springs was DDE at low concentrations, near the LRL, in two of 13 samples. The occurrence of DDE in discharging spring sediment confirms that surface-water sediment is transported through the aquifer, but the time scale for this transport is unknown—discharging sediment could have infiltrated the aquifer during the sampled event or at an earlier time. The very low DDE concentrations in spring sediment compared to those in creek sediment indicate that contaminated sediment in recharge is diluted by "cleaner" sediment. The origin of this cleaner sediment could be recharge from less-developed watersheds such as Onion Creek or Bear Creek, historical recharge from creeks decades ago (before the use of organochlorine compounds), erosion from within the aquifer itself, or a combination of these sources.

Concentrations of PAHs at Barton Springs are about the same as those at the next least contaminated site (Barton 71), on the basis of median total PAH (table 7). Again, it is not known if PAH concentrations are low because more contaminated, recently recharged sediment has not yet been transported to the springs, or because contaminated sediment in recharge is being diluted by cleaner sediment. Contaminated sediment infiltrating the aquifer might go into storage to be resuspended and moved farther through the system during each event or during periods of higher ground-water levels, when ground-water velocities are faster (Hauwert and others, 2001).

No statistically significant trends in contaminants were associated with sediment discharging from Barton Springs for the 5 years monitored, although the effect of urbanization on water quality at Barton Springs has been seen for other constituents over longer time scales. An analysis of trends in water quality from Barton Springs by the City of Austin found statistically significant upward trends in specific conductance, turbidity, sulfate, and total organic carbon and a downward trend in dissolved oxygen from 1975 to 1999 (Turner, 2000). Water samples collected from the Barton Springs system by the USGS and analyzed for polar pesticides have detected as many as seven different pesticide compounds; all seven also were detected in Williamson and Slaughter Creeks, and six were detected in Barton Creek (Mahler and others, 2006). The analytical method targets pesticides associated with the water phase.

The periodic overtopping of the upstream dam that separates Barton Creek from Barton Springs pool represents an immediate pathway for the delivery of contaminated sediment to the pool. Flow in Barton Creek at Barton Above normally is routed around Barton Springs pool through a bypass tunnel; however, when flow in the creek reaches a threshold of about 500 ft^3/s (less when the opening to the bypass is blocked by debris), the creek overtops the dam and flows into the pool. This can occur as often as 2 or 3 times a year, or it might not occur for a year or more, depending on weather conditions. Each time the dam is overtopped, sediment from Barton Creek—the same sediment that is sampled at Barton Above—is deposited in a visible layer on the bottom of the pool. It can range in thickness from a thin veneer of silt less than 1-in. thick to gravel bars 1-foot or more thick. Sediment transported by rain events in the downstream part of Barton Creek is much more contaminated than that sampled in the upstream part of the watershed or that discharging from Barton Springs (figs. 6, 8, 9, 17; appendixes 3–5). In terms of the health of aquatic biota living in the pool, such as the Barton Springs salamander, sediment transported by stormflow probably represents a more immediate concern than sediment discharging from the aquifer.

Within-Event Variation in Concentrations and Loads—Case Studies for Shoal and Boggy Creeks

Within-event variation in contaminant concentrations and loads was investigated for Shoal and Boggy Creeks (sites Shoal and Boggy) by collecting and analyzing individual samples distributed in time over the storm duration. These data were used to answer several questions: Which, if any, hydrophobic contaminants show a first-flush effect, representing undiluted runoff from nearby impervious surfaces? Do contaminants of the same type (metals, organochlorine compounds, and PAHs) show similar temporal patterns? Can temporal patterns provide clues about distance of travel? How does temporal variation in contaminant concentrations compare to that of contaminant loading? Are the temporal patterns consistent from one site to the next? The answers to these questions might be useful in identifying sources of contaminants and in improving approaches for mitigating contamination.

Among the sites investigated for this study, Shoal and Boggy are the two with the most urban (developed) land use, more than 80 percent in both cases (table 1). Although both watersheds are highly urbanized, the amount and spatial distribution of urban land uses are somewhat different. The Shoal Creek watershed is characterized by commercial development in the part of the watershed nearest the gaging station, single-family housing in the middle part of the watershed, and commercial, multifamily housing, and transportation land use in the upper part of the watershed (fig. 21A). In contrast, the Boggy Creek watershed is characterized by more warehousing and industrial land use in the lower part of the watershed and more residential land use in the upper part of the watershed (fig. 21B).

Individual discrete samples over the storm duration were analyzed for four rain events at Shoal and two at Boggy (appendix 6). The hydrographs of the events sampled, the times represented by the discrete samples, and the corresponding SSCs are shown in figure 22. For the first event at Shoal, March 18, 1999 (SC1, fig. 22A), the first flush was missed and only four samples were collected. However, a sample was collected during the first flush of a very small event that occurred 6 days earlier (March 12, 1999). This sample was not included in the composite calculation for the March 18 rain event, but it is shown in the figures for this section with the rest of the discrete samples for event SC1 to represent the first-flush sample. For Shoal events SC2 (April 26, 1999), SC3 (October 30, 1999), and SC4 (March 17, 2000) (fig. 22A) and for Boggy events BC1 (October 15, 2000) and BC2 (January 1, 2001) (fig. 22B), samples were analyzed either individually or combined into flow-weighted composites such that for each event five samples were analyzed. The events sampled represent a range of sizes, but in general, the samples that were analyzed represent the first flush, a point on the rising limb of the hydrograph, a point at or near cresting stage, and two points on the falling limb of the hydrograph.

At Shoal, each of the hydrographs shows a characteristic small shoulder early in the event (first 2 hours after water begins to rise), which then is followed by a much larger peak. The shoulder could represent local runoff from the commercial area and roads near the gage. The first sample usually was collected from this shoulder (fig. 22A). At Boggy, event BC1 has a single peak, and BC2 has a double peak and a much longer duration (fig. 22B), probably resulting from a bimodal distribution of rainfall that day. The results of analyses of major elements, metals, organochlorines, and PAHs are in appendix 6.

Temporal Patterns in Concentrations Over the Duration of the Storm

Major element chemistry indicates that variations in the mineralogy of suspended sediment in Shoal and Boggy Creeks are different over the course of rain events (fig. 23). At Shoal, Al and Fe concentrations, which reflect the clay mineral component, both increase initially then plateau, whereas Ca concentrations, which reflect the carbonate component, are relatively low in the first-flush sample, peak early in the event, and then decrease. These patterns indicate that at Shoal, during the latter one-half of the event, the clay fraction remains high and the carbonate fraction decreases. This could be a reflection of stream power: When the creek is cresting, turbulence and stream power are sufficient to keep the larger carbonate particles in suspension, whereas later in the event mostly smaller, platey minerals (clays) remain in suspension. The grain-size distribution is consistent with this hypothesis, with the greatest proportion of silt- to sand-sized particles (3 to about 30 percent) in the second or third sample collected from each event (appendix 6). In contrast, at Boggy, the Al, Fe, and Ca concentrations remain relatively constant over the course of both storms sampled, indicating little variation in the mineralogy of the suspended sediment. The Fe concentration at Boggy is about 5 times that at Shoal, likely reflecting geologic differences between the two watersheds.

Temporal patterns of Na over the course of rain events at Boggy and Shoal are similar, whereas temporal patterns of P are different. At Shoal, concentrations of both Na and P, in contrast to Al and Ca, are highest in the first sample collected from each event (fig. 23), which indicates a surface-runoff origin. Similarly, at Boggy, the Na concentration was elevated in the first sample collected from both events, although it was much higher in the first sample from event BC1 compared to that from event BC2. In contrast, concentrations of P in samples from Boggy varied little over the course of either storm event. Although Na and P are common in sedimentary rocks, a difference between their behavior and that of Al and Fe suggests that in some cases they also have an anthropogenic source. Irrigation and landscape water, particularly in arid climates, can leave Na-rich residual at the surface (Hem, 1985). Phosphate fertilizers added to soils can, through erosion, greatly increase the amount of suspended P in streams (Hem, 1985). Atmospheric deposition also could contribute to P concentrations on surfaces that are then washed off by rain. Differences in the temporal variation in P over the course of rain events between the two watersheds suggests that Shoal Creek may be more affected by use of P-rich fertilizers than Boggy Creek.

Temporal patterns in metals concentrations differed among metals and among sites. At Shoal, concentrations of Cd, Cu, Hg, Pb, and Zn were much higher in the first sample collected from the storm, decreased greatly, and then varied little over the remainder of the storm (fig. 24), similar to the patterns seen for Na and P at this site. In the most extreme cases, such as for event SC2, the concentration in the first sample is as much as 5 times higher than the concentration in the next sample collected. The elevated concentrations in the first sample collected, followed by a rapid decrease, reflect the “first flush” effect: Those contaminants that build up on impervious surfaces (such as roadways, parking lots, and rooftops) that provide an avenue for rapid delivery to storm sewers and urban creeks are elevated in the sample or samples collected early in the storm response. The elevated concentrations of these five metals in the first flush thus indicate an anthropogenic source. At Boggy, in contrast, no first flush of Cd, Cu, Hg, Pb, and Zn was seen. Concentrations of these metals varied little over the course of the storm, with the exception of one sample of Cd collected in the middle of event BC2.

Temporal patterns in As, Cr, and Ni were different from those of Cd, Cu, Hg, Pb, and Zn. Concentrations of As varied little over the course of the rain events, with the exception of an elevated concentration in the first sample collected at Boggy during event BC2. At Shoal, Cr and Ni both had a first-flush effect, indicating an anthropogenic source for these metals at the beginning of the event. After decreasing, however, the concentration then increased during the second part of most events, in many cases to a concentration exceeding the concentration in the first sample. The increase in concentration during the second part of the event is similar to that of Al at this site, and indicates that the increase might result from the contribution of a mineralogic, non-anthropogenic source. At Boggy, Cr varied little over the course of either rain event, similar to the behavior of Al. Ni, however, increased greatly over the course of event BC1 and decreased over the course of BC2; the difference

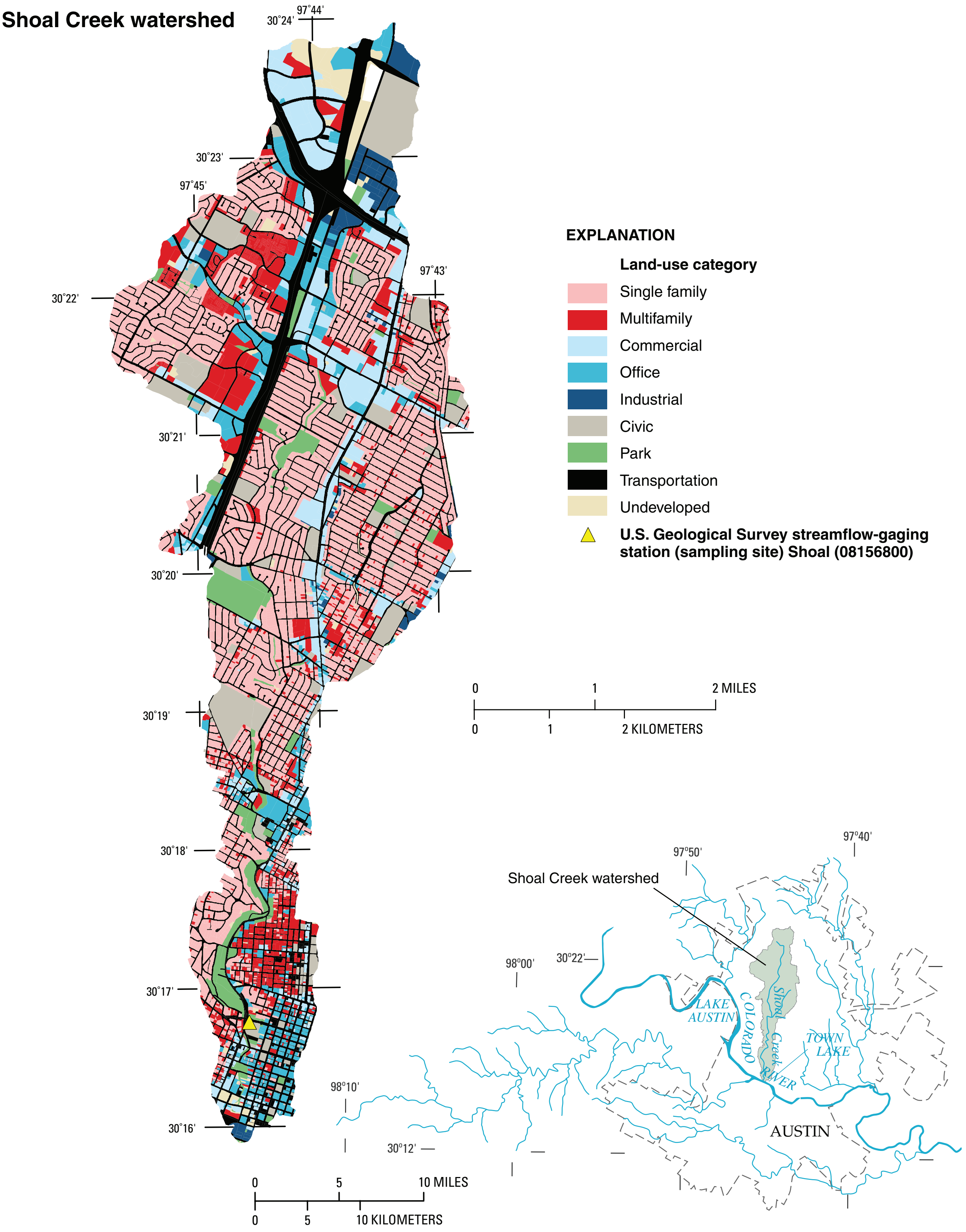


Figure 21A. Land use in Shoal Creek watershed, Austin, Texas, 2003 (Robert Botto, City of Austin, written commun., 2003).

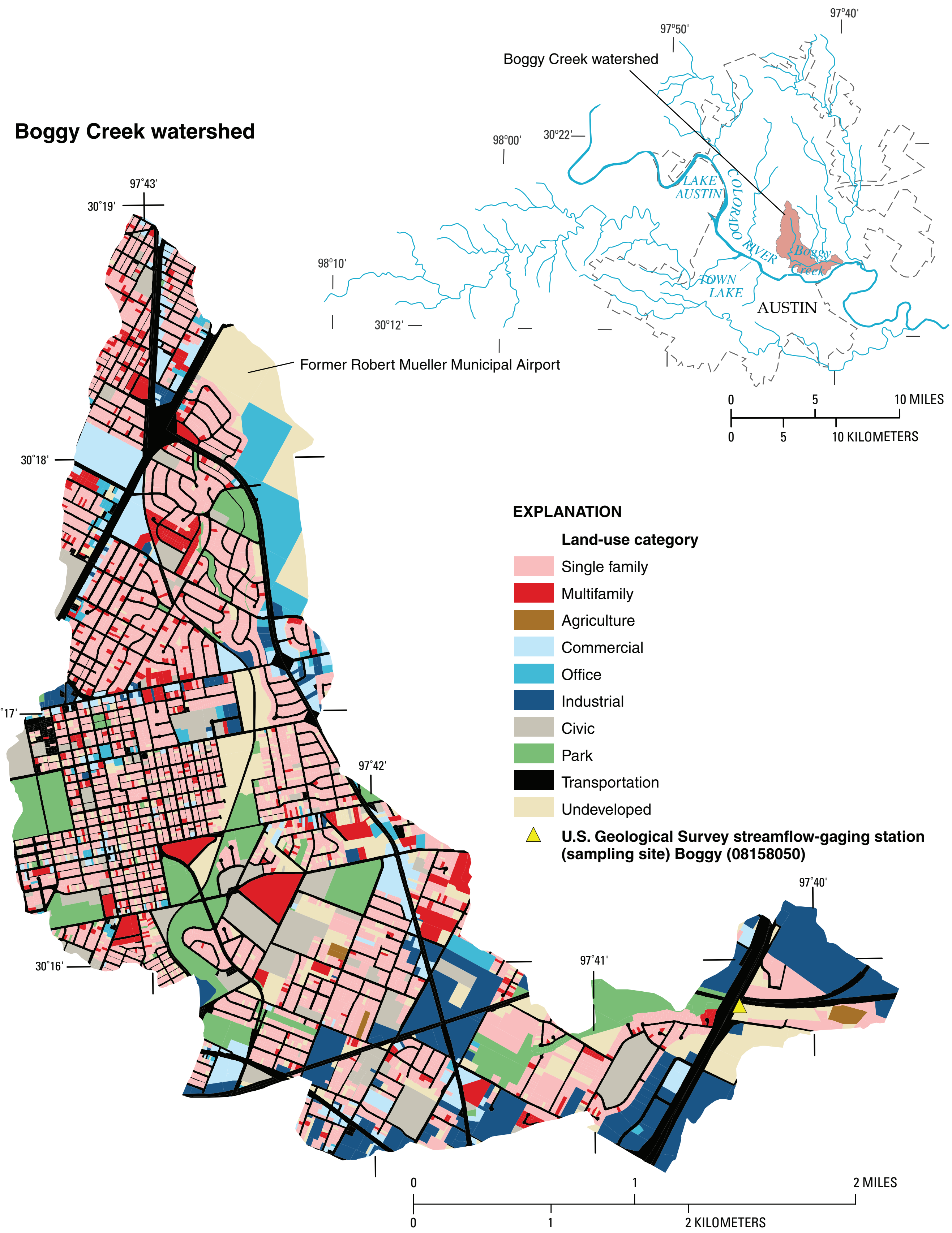


Figure 21B. Land use in Boggy Creek watershed, Austin, Texas, 2003 (Robert Botto, City of Austin, written commun., 2003).

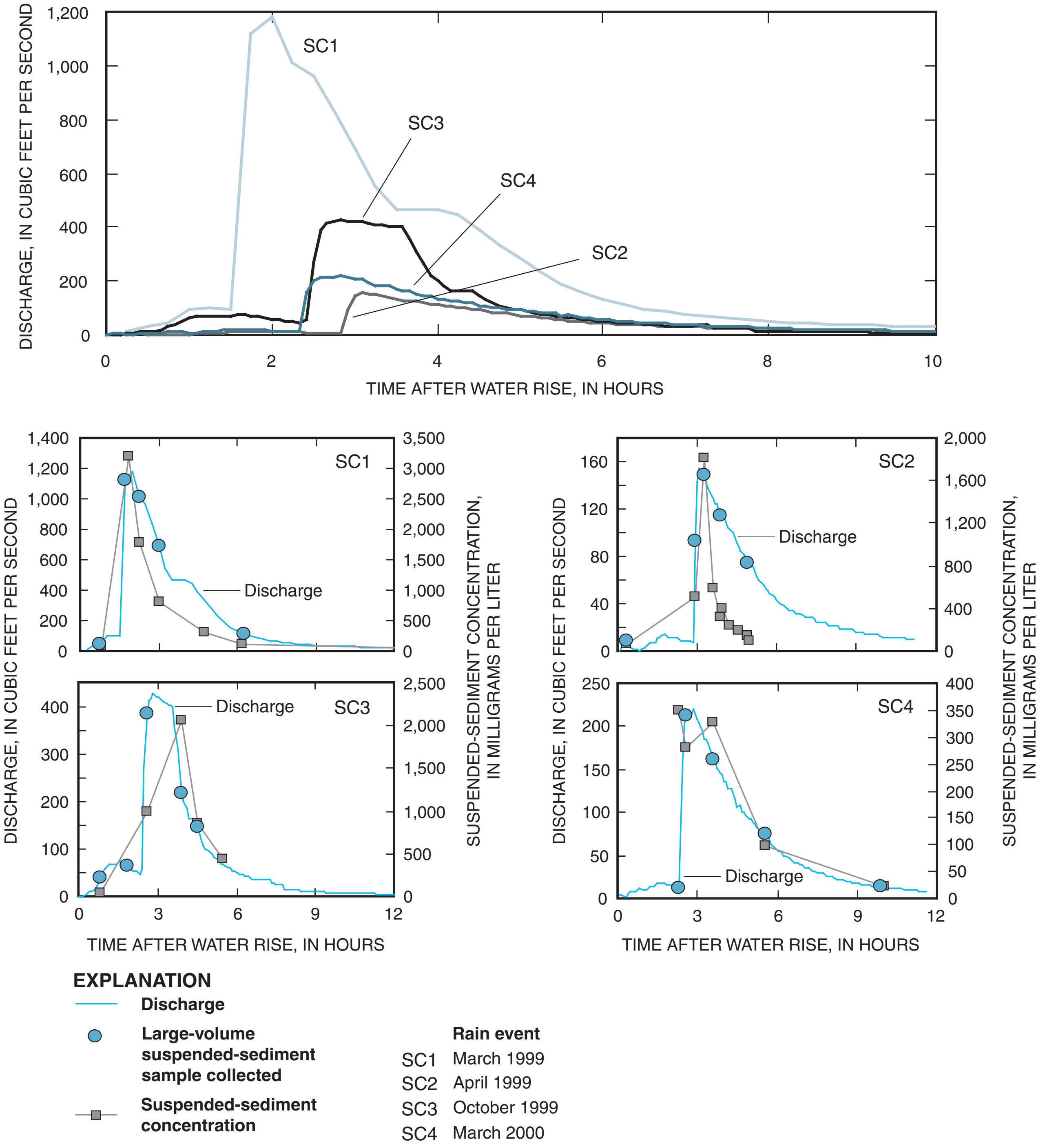


Figure 22A. Discharge hydrographs corresponding to discrete samples and suspended-sediment concentration for rain events at Shoal Creek, Austin, Texas, 1999–2000.

between the behavior of this metal and that of Al indicates that in the Boggy Creek watershed Ni might have an anthropogenic source.

Overall, variations of concentrations of metals at Boggy do not indicate that the source of the metals is anthropogenic. Metals concentrations vary similarly to those of major elements associated with clays and show little to no first-flush effect. In most cases, concentrations of metals at Boggy throughout the event are similar to those at Shoal at the end of the event when the first-flush effect has subsided.

Patterns in concentrations of organochlorine compounds vary from event to event (fig. 25). In some cases at Shoal,

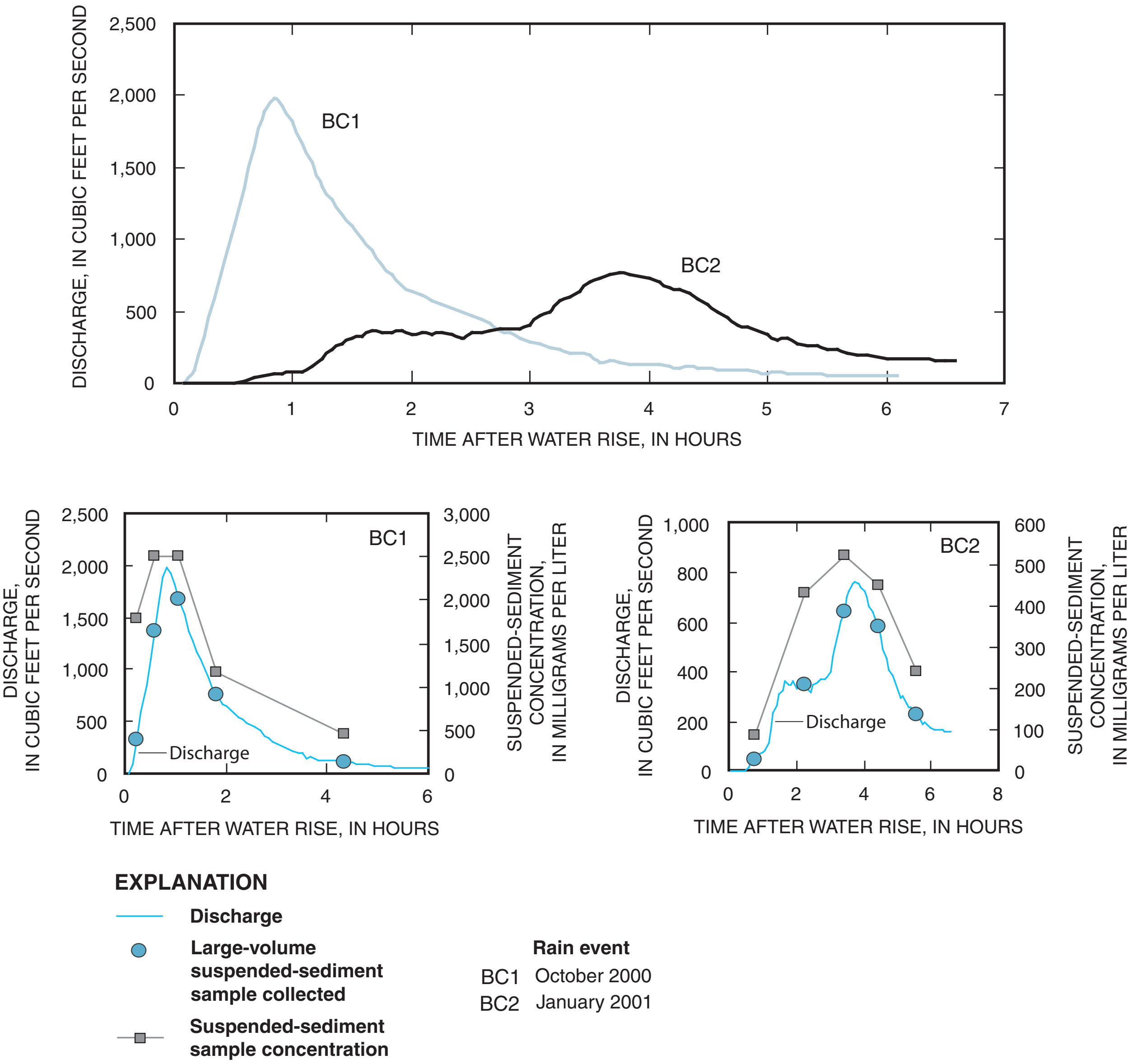


Figure 22B. Discharge hydrographs corresponding to discrete samples and suspended-sediment concentration for rain events at Boggy Creek, Austin, Texas, 2000–2001.

patterns are similar to those seen for metals: an elevated concentration in the first sample, followed by a decrease, and then a small increase in concentration toward the end of the event (for example, DDE in SC1, DDT in SC4). For SC2, the first sample was collected within about 30 minutes of the water rise, and the concentration of DDE was low and DDT was not detected. For events SC3 and SC4, the highest concentrations of DDT were in the first samples collected, but these were collected about 50 minutes and 2.5 hours after water rise. Dieldrin and chlordane were detected in at least one of the samples from three of the four events sampled at Shoal. The highest concentrations were in the first sample for every event for which dieldrin and chlordane were detected; in most cases concentrations decrease monotonically throughout the event, usually with nondetections in the final samples. PCBs were detected only for SC4 but have a similar pattern. These patterns indicate that the sources of dieldrin and chlordane, and possibly PCBs, are near the gaging station in the Shoal Creek watershed.

At Boggy, concentrations of chlordane, DDE, and PCBs are elevated in the first sample collected for event BC1 but were either relatively low or not detected in the first sample collected for BC2. This might occur because the first sample for BC1 was collected 15 minutes after the start of the water rise, and the first sample for BC2 was collected 45 minutes after the start of the water rise. On the basis of timing, concentrations of chlordane, DDE, and PCBs are quite similar for the two events. The results indicate that sources of these three organochlorine compounds are distributed throughout the watershed.

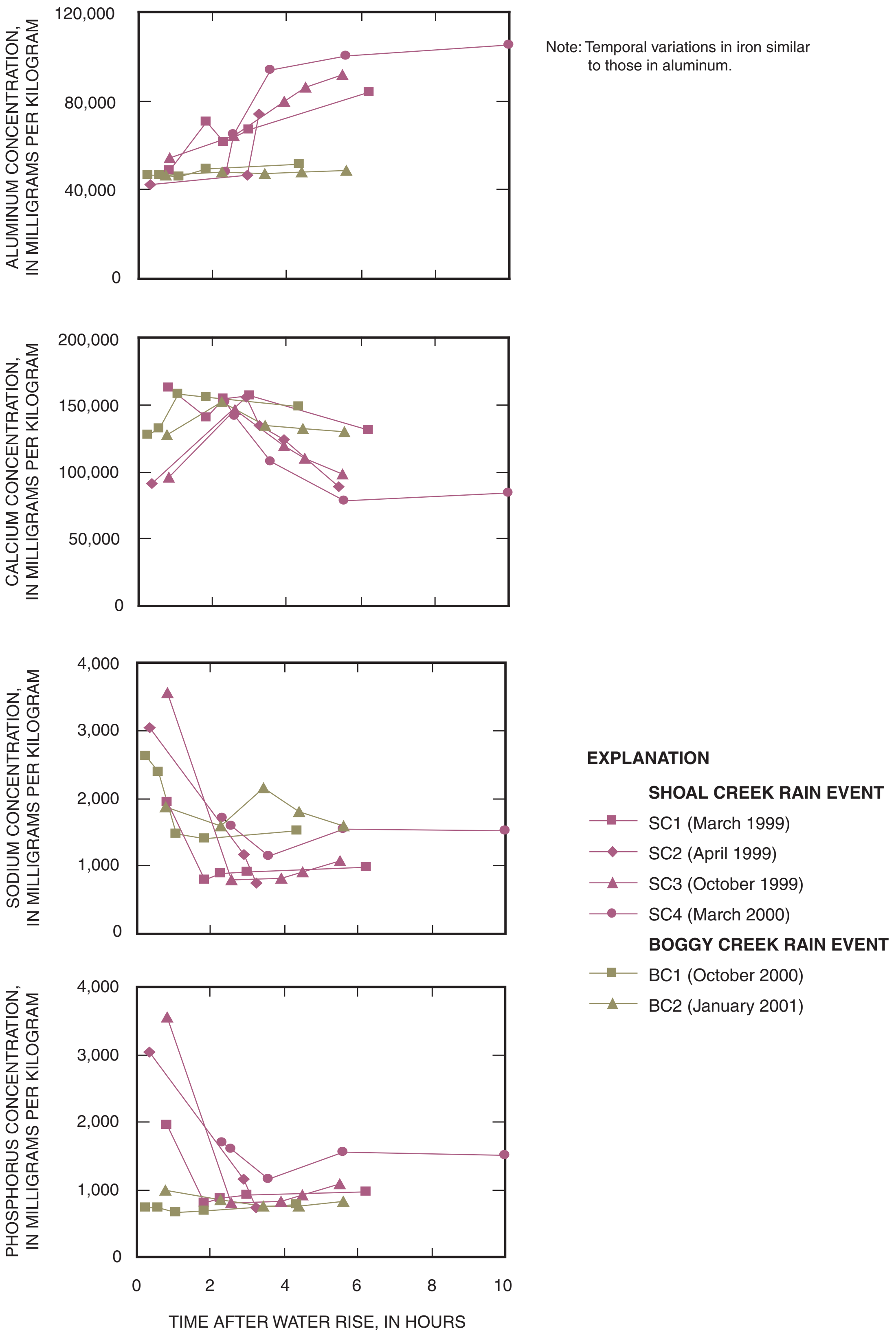


Figure 23. Concentrations of selected major elements in discrete suspended-sediment samples collected during rain events at Shoal and Boggy Creeks, Austin, Texas, 1999–2001.

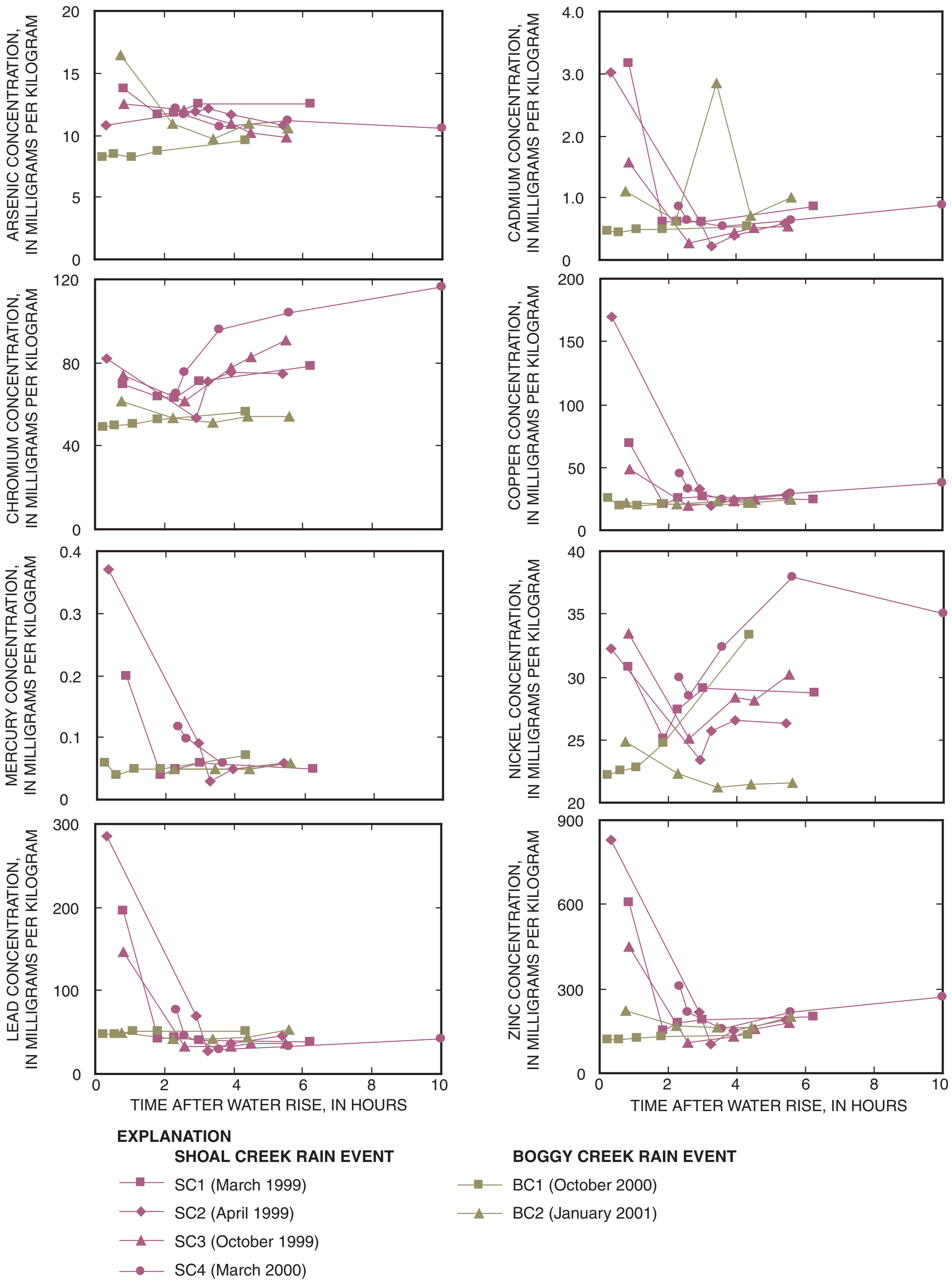


Figure 24. Concentrations of metals in discrete suspended- sediment samples collected during rain events at Shoal and Boggy Creeks, Austin, Texas, 1999–2001.

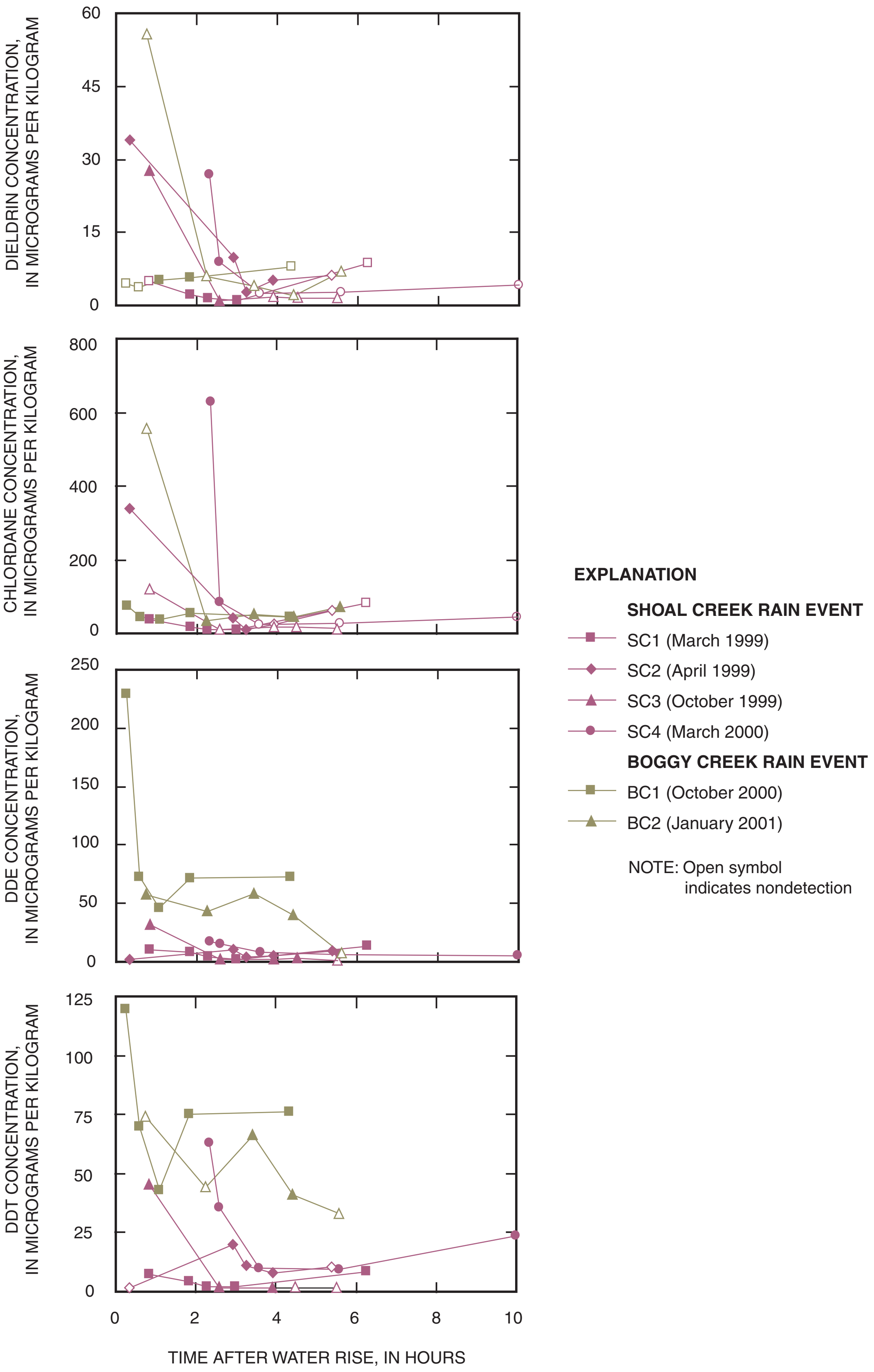


Figure 25. Concentrations of organochlorine compounds in discrete suspended-sediment samples collected during rain events at Shoal and Boggy Creeks, Austin, Texas, 1999–2001.

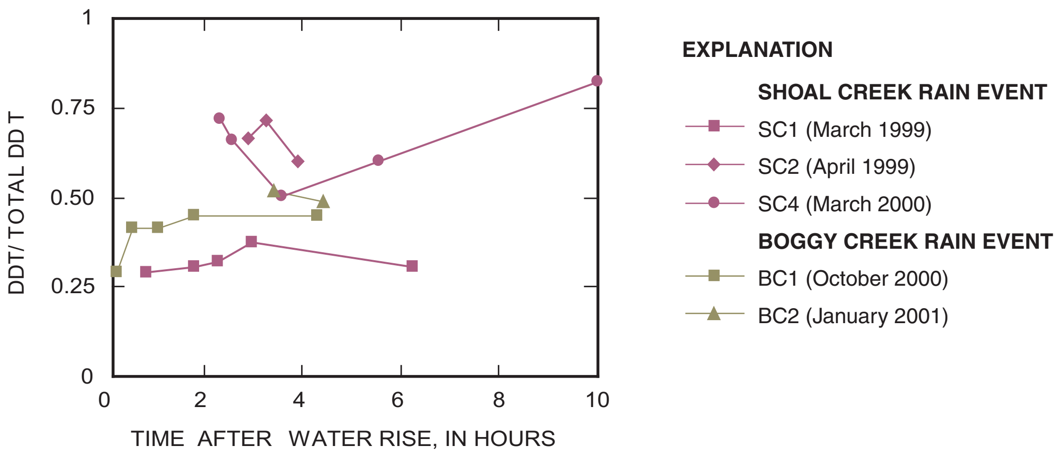


Figure 26. Changes in ratio of DDT to total DDT in discrete suspended-sediment samples collected during rain events at Shoal and Boggy Creeks, Austin, Texas, 1999–2001.

Temporal patterns in the ratio of parent DDT to total DDT are dissimilar for the two watersheds (fig. 26). This is of interest because the ratio of DDT to total DDT is thought to indicate how recently the DDT has been mobilized—the higher the ratio, the more recently the DDT has been introduced into the hydrologic system (Nowell and others, 1999). Ratios at Shoal show more variation both within and between events than do ratios at Boggy. This indicates that multiple sources of soil contaminated with DDT are in the Shoal Creek watershed, some of which have been mobilized more recently than others. The relatively constant ratio in Boggy, in combination with little change in concentration during events, indicates that the source of DDT might be streambed soils that continue to erode.

Temporal patterns in PAH, as typified by ΣPAH_{SQG} concentrations, vary greatly between Shoal and Boggy (fig. 27). At Shoal, temporal patterns generally are similar to those of the metals (fig. 24), which indicates a similar source or source type. For three of the four events (SC2–SC4), the concentration is highest at the beginning and then decreases, with a slight increase toward the end of the event. For these same events, the source-indicator ratio (ratios of 2- and 3-ringed compounds to combustion compounds) remains low and constant, from 0.1 to 0.2, which indicates a combustion source for PAHs in all samples.

At Boggy, ΣPAH_{SQG} concentrations for event BC1 were relatively low and unchanging, whereas concentrations for BC2 increased throughout the event, and the maximum concentration reached at the end of the event was about 3.5 times higher than the concentration of the initial sample (fig. 27). The source-indicator ratio for BC1 was relatively constant, from 0.1 to 0.2, similar to ratios for Shoal. The source-indicator ratio for BC2, in contrast, decreases from about 0.4 to 0.1 as concentrations of ΣPAH_{SQG} increase from 4,000 to 14,000 µg/kg over a 6-hour period. This indicates that the increase in PAH concentration results from a source with a strong combustion signal that originates fairly far up the watershed. One possible source of these PAHs is the former Robert Mueller Municipal Airport, in operation from 1930 to 1999 and in the upper northeastern part of the watershed (fig. 21B), although there is a large amount of office and commercial development in the upper part of the watershed as well.

Temporal Patterns in Loads

Temporal loads data describe the proportion of total event load that was in transport during the event. Each data point refers to the total mass of a constituent passing the sampling point during the interval of time represented by the sample analyzed. The data were computed by multiplying the mass of contaminant per liter of water (the product of sample concentration [mass of constituent per mass of sediment] and SSC [mass of sediment per volume of water]) by the total volume of water that passed the gaging station during the time interval represented by the sample.

For the events analyzed, temporal variations in loads generally follow the same form as the hydrograph (fig. 28), rather than showing the first-flush effect seen in the concentrations. The marked differences between the load and concentration patterns arise from the fact that contaminant load is a function of sediment load (that is, contaminant concentrations are multiplied by sediment load to obtain contaminant load) and concentrations are not. Sediment load, in turn, is a function of SSC and streamflow, both of which peak at about the same time (fig. 22). Decreases in concentration in the middle of the rain event are more than compensated for by the increased mass of sediment in transport, which results in a higher load. In contrast, the first part of the hydrograph, although containing the most highly contaminated sediment, does not contribute much to the load because both streamflow and SSC are relatively low. Because the contaminant load largely is governed by SSC and streamflow rather than by concentration, temporal variations in loads for the different contaminants for a given event are very similar.

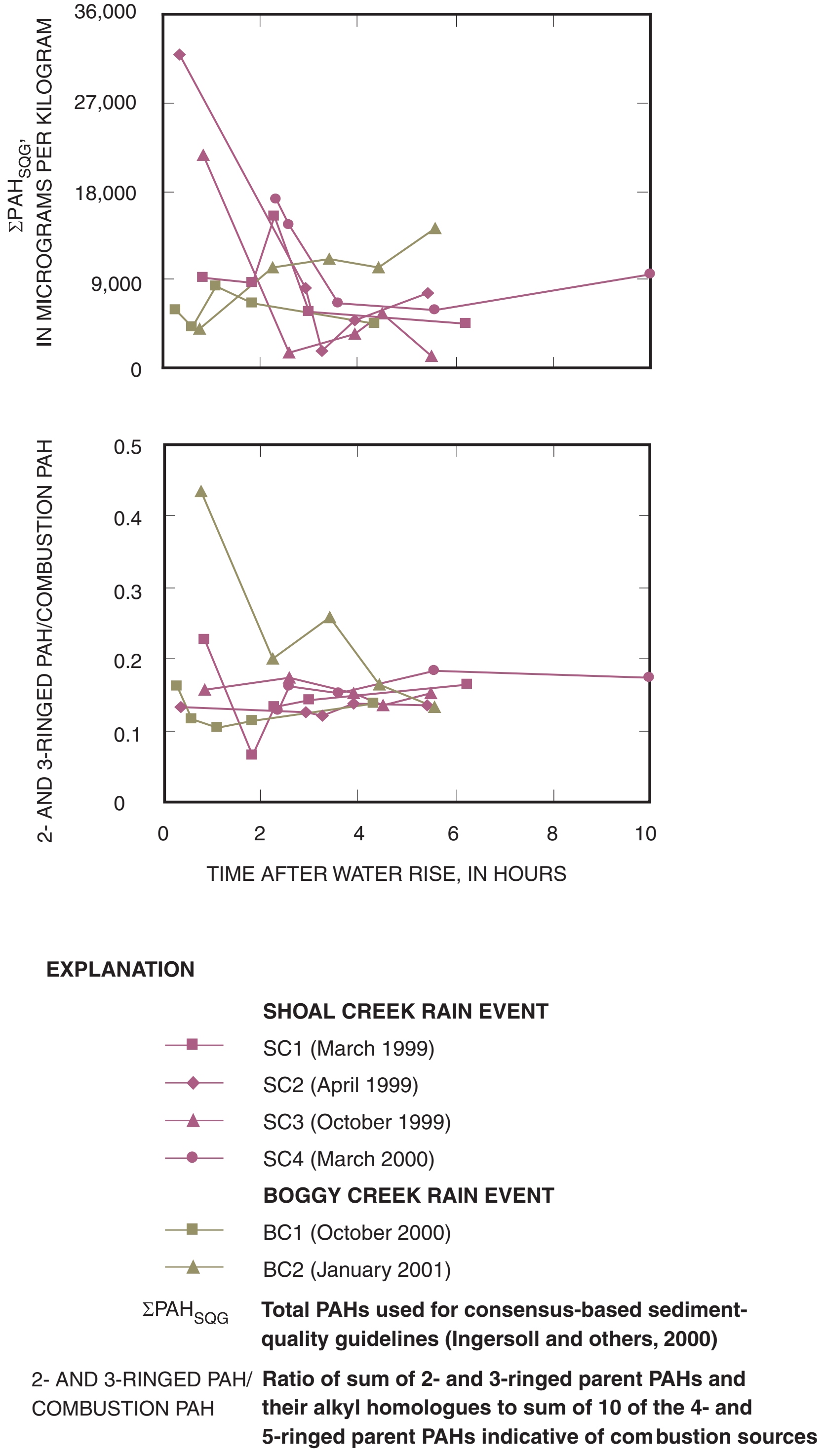


Figure 27. Concentrations of polycyclic aromatic hydrocarbons (PAHs) and changes in source-indicator ratio of PAHs in discrete suspended-sediment samples collected during rain events at Shoal and Boggy Creeks, Austin, Texas, 1999–2001.

Table 8. Measured dissolved metals concentrations in streamflow during three rain events at Shoal Creek, Austin, Texas, 1999.

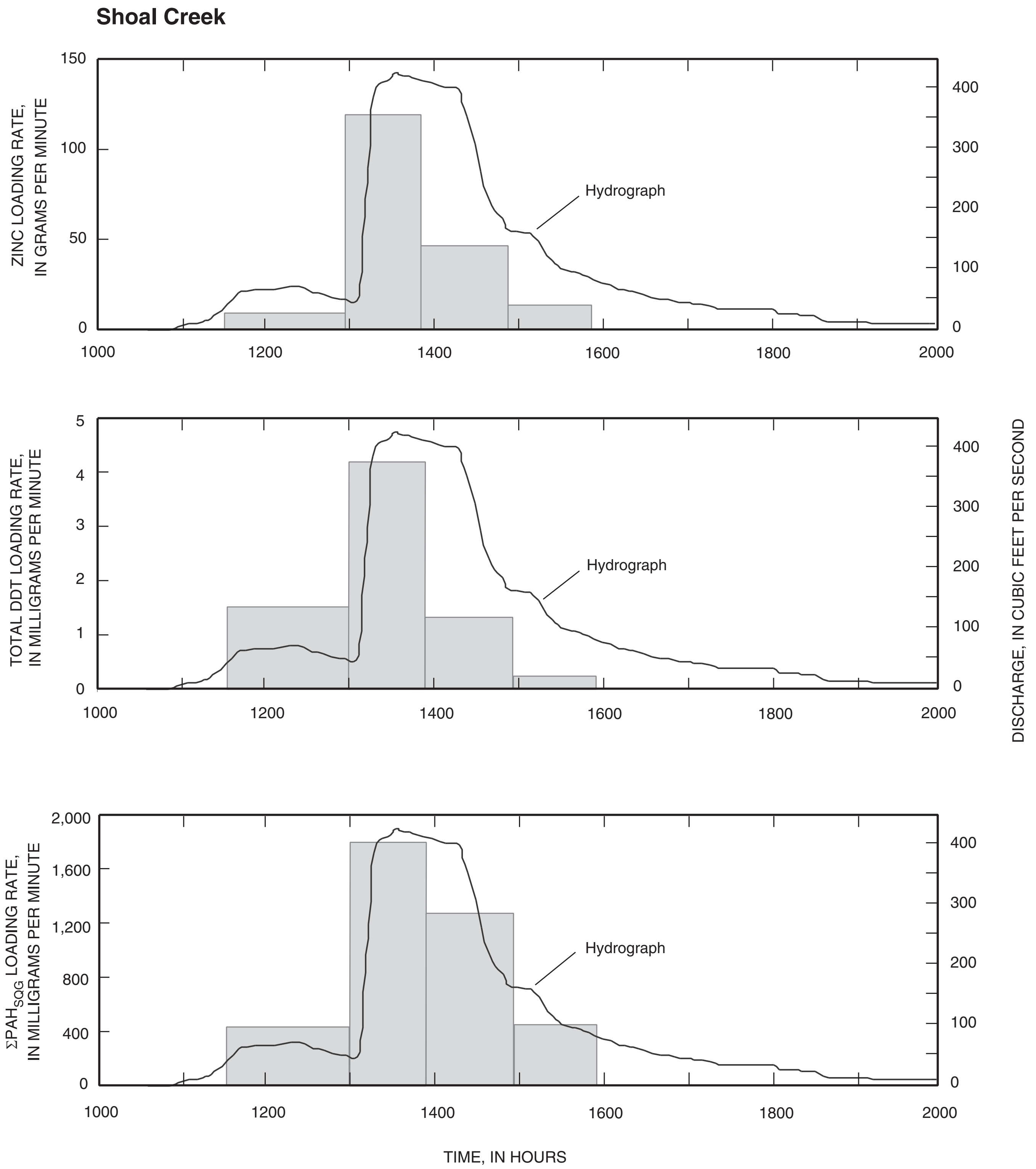
[SSC, suspended-sediment concentration; mg/L, milligrams per liter; µg/L, micrograms per liter; <, less than; --, not analyzed].

Rain event	Sampling date	Time	SSC (mg/L)	Cadmium, dissolved (µg/L)	Chromium, dissolved (µg/L)	Copper, dissolved (µg/L)	Nickel, dissolved (µg/L)	Lead, dissolved (µg/L)	Zinc, dissolved (µg/L)
SC1	03/12/1999	1200	83	1.1	<1	6.6	2.7	5.9	22.9
	03/18/1999	2120	3,181	1.0	--	1.2	2.6	1.6	21.1
		2147	1,774	<1	1.2	1.7	3.2	1.6	40.1
		2230	819	<1	<1	1.7	1.7	<1	31.2
		0144	116	<1	<1	1.3	1.2	<1	26.7
SC2	04/26/1999	0910	59	<1	--	9.0	3.5	5.2	81.5
		1145	504	<1	1.4	2.8	3.5	<1	31.4
		1205	1,813	<1	<1	1.6	4.0	<1	79.8
		1247	390	<1	<1	2.3	5.5	<1	99.5
		1347	96	<1	<1	2.5	3.7	<1	13.5
SC3	10/30/1999	1030	31	<1	<.8	5.2	2.7	1.9	29.6
		1215	983	<1	2.2	2.8	7.3	<1	29.0
		1335	2,066	<1	1.0	2.5	5.5	<1	7.1
		1410	845	<1	1.0	3.0	6.0	<1	8.5
		1510	443	<1	1.0	2.6	4.1	<1	5.6

Temporal Patterns in Contaminant Partitioning—Comparison of Particulate and Dissolved Metals Concentrations

Concentrations of metals in the dissolved and particulate phases were compared to see how the fraction of the total contaminant mass in each phase changed over the hydrograph for three of the four rain events sampled at Shoal. Concentrations of six metals (Cd, Cr, Cu, Ni, Pb, and Zn) in the dissolved phase were analyzed for events SC1–SC3 (table 8). In most cases the filtration was done within 24 hours of sample collection; however, even within this time frame there could have been some loss of contaminant from the solid phase to the dissolved phase, so the concentrations of metals associated with the dissolved phase should be considered maximums. The concentration associated with the particulate phase on a volumetric basis (rather than on a mass basis) was computed so that it could be compared to that in the dissolved phase. This was done by multiplying the concentration (milligrams per kilogram) analyzed for in the suspended sediment by the SSC (milligrams per liter) and dividing by 1,000 to obtain micrograms of contaminant associated with the particulate phase per liter of water. For each metal the proportion of the total load for each event associated with suspended sediment was computed. For those metals with nondetections in the dissolved phase (Cd, Cr, and Pb), a maximum and a minimum proportion associated with suspended sediment were computed, with the maximum corresponding to a zero concentration for nondetections in the dissolved phase, and the minimum corresponding to a concentration equal to the minimum reporting level (MRL) for the nondetections in the dissolved phase. The only metal for which this resulted in a wide range in proportions is Cd, which has a fairly high MRL for the dissolved phase relative to concentrations detected on the particulate phase. There were numerous nondetections for Cd in the dissolved phase (13 of 15 samples, or 87 percent).

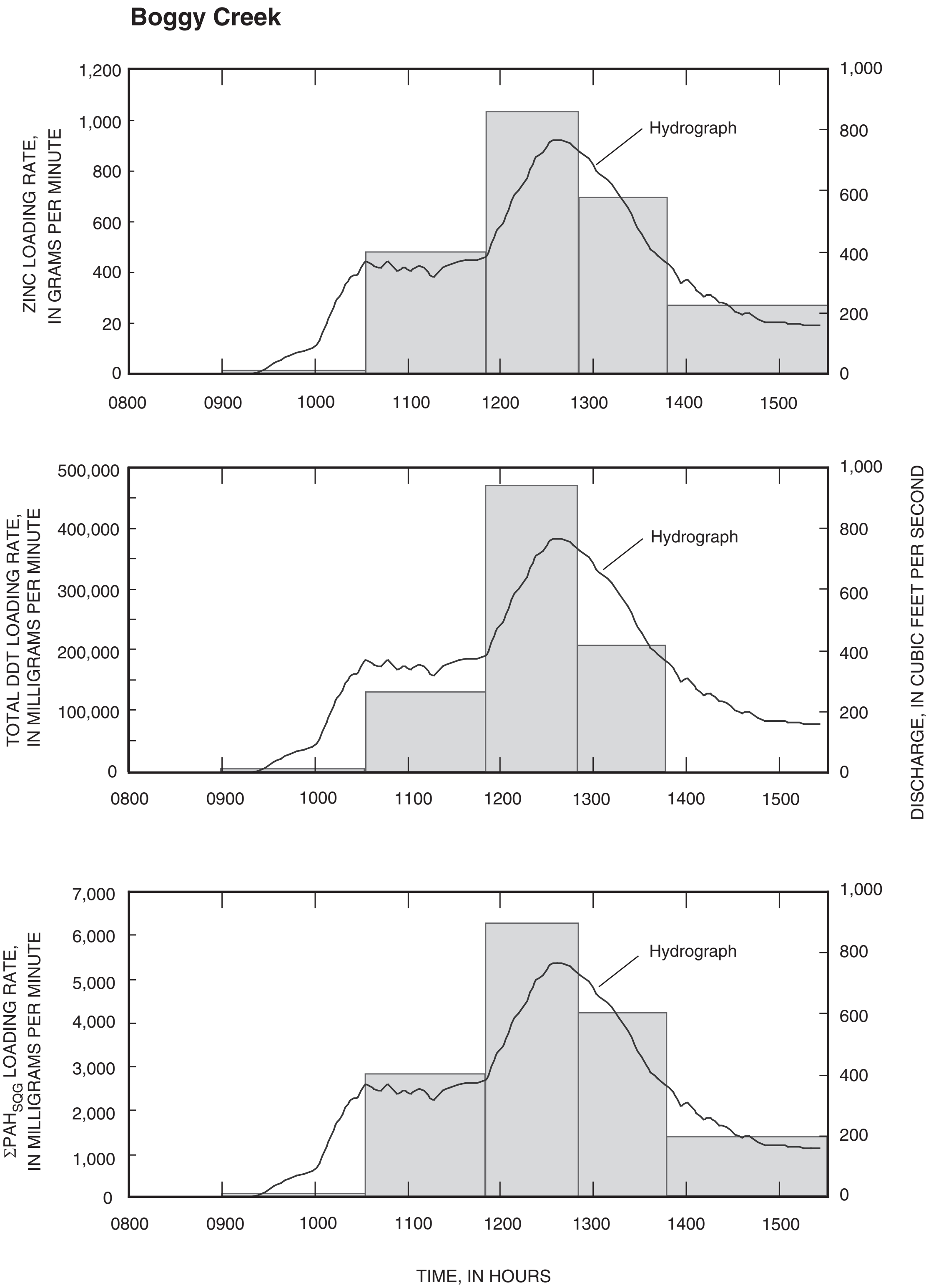
Most of the mass of each metal in a liter of water was associated with the suspended sediment phase (fig. 29). In many cases, there was no detectable concentration in the dissolved phase: 30 of the 88 dissolved-phase analyses (34 percent) were nondetections, all for Cd, Cr, and Pb. For all samples combined, the median proportion of an individual metal associated with sediment was between 79 percent (assigning the nondetections a concentration equal to the MRL) and 92 percent (assigning the nondetections a zero concentration). On the basis of the medians for each metal, the metals with the highest proportions associated with sediment were Cd and Cr (greater than 99 percent of the metal mass associated with sediment if nondetections are counted as zero), and the lowest were Ni and Zn (77 and 78 percent associated with sediment). In general, the proportion of mass associated with the suspended phase tended to be lowest when the SSC was low (beginning [sample 1] and end [sample 5] of the event) (fig. 29), and that amount makes up only a small part of the total contaminant load. Thus for these rain events, the proportion of the metals associated with the particulate phase makes up the major part of the total load (fig. 30), and particle-associated loads of metals computed using LVSS data can provide estimates of total loads.



EXPLANATION

ΣPAH$_{SQG}$ **Total PAHs used for consensus-based sediment-quality guidelines (Ingersoll and others, 2000)**

Figure 28A. Temporal variations in loads of zinc, total DDT, and polycyclic aromatic hydrocarbons (PAHs) during a representative rain event at Shoal Creek, Austin, Texas.



EXPLANATION

ΣPAH_{SQG} **Total PAHs used for consensus-based sediment-quality guidelines (Ingersoll and others, 2000)**

Figure 28B. Temporal variations in loads of zinc, total DDT, and polycyclic aromatic hydrocarbons (PAHs) during a representative rain event at Boggy Creek, Austin, Texas.

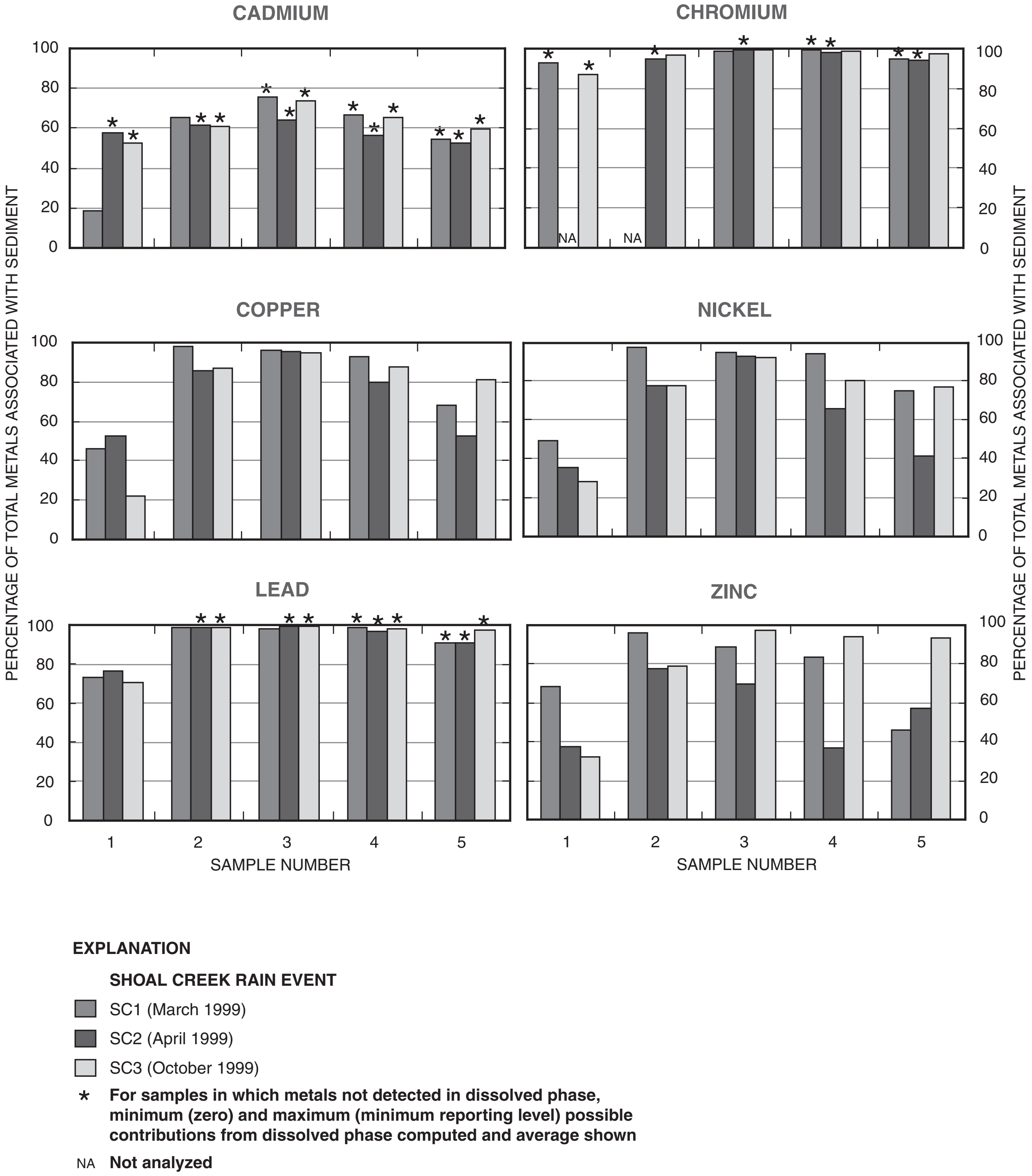


Figure 29. Temporal patterns of partitioning of metals between dissolved and suspended phases for rain events SC1, SC2, and SC3 at Shoal Creek, Austin, Texas, 1999.

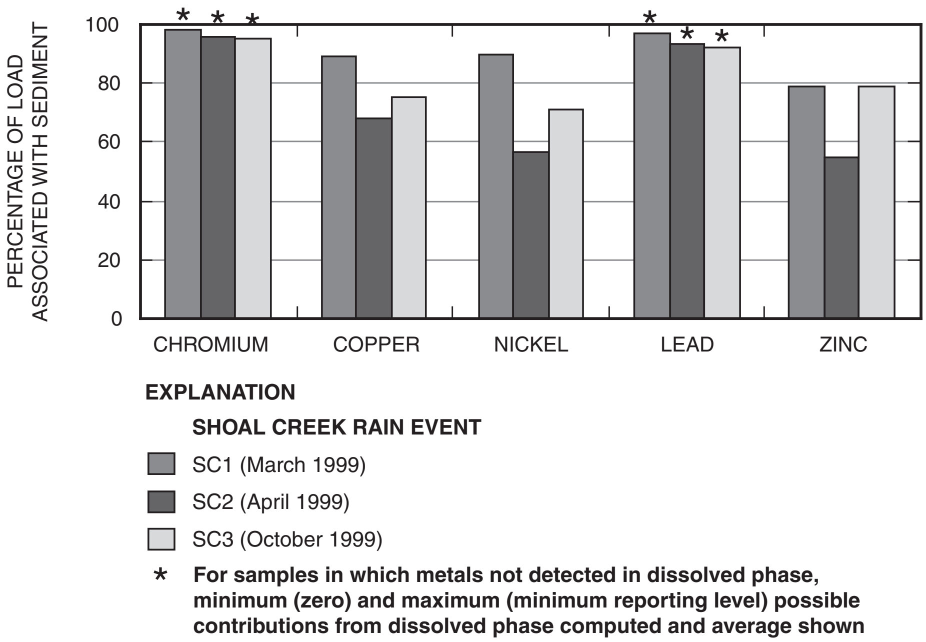


Figure 30. Proportion of total metal load associated with suspended sediment for rain events SC1, SC2, and SC3 at Shoal Creek, Austin, Texas, 1999.

Conclusions

Concentrations, loads, and yields of hydrophobic contaminants were investigated in some Austin, Tex., urban creeks using LVSS—large-volume suspended-sediment sampling—in a study by the U.S. Geological Survey, in cooperation with the City of Austin. Specifically, concentrations of major elements, metals, organochlorine compounds, and PAHs associated with particles in stormflow in Barton, Shoal, Boggy, and Williamson Creeks, and in Barton Springs were sampled and interpreted for selected rain events from March 1999 through October 2004. The major conclusions of the study are

- *Different creeks are affected by different contaminants, but in general suspended sediment at the more urban sites have elevated concentrations of hydrophobic contaminants.* The highest concentrations of metals were detected in Shoal and Williamson Creeks, the highest concentrations of organochlorine compounds in Shoal Creek, and the highest concentrations of PAHs in Williamson Creek. Suspended sediment from sampling sites Shoal and Williamson had elevated contamination indexes (C_I): The median C_I from Williamson exceeded 0.5, the threshold in bed sediment for expected adverse effects on benthic biota, and the median C_I for Shoal was 0.47. The least urbanized watershed (upstream from Barton 71) produced relatively uncontaminated runoff, with a median C_I of 0.07. The median C_I at Barton Springs also was very low (0.12). At the more contaminated sites, organochlorine compounds and PAHs contributed much more to sediment contamination than metals.
- *Loads of almost all contaminants were correlated with sediment load.* A statistically significant relation occurred because of much more between-event variability in the suspended-sediment load than in the contaminant concentrations. As a result, once a characteristic contaminant concentration has been determined for a site, contaminant loads and yields for individual events can be estimated from measured sediment loads, provided that the contaminant concentration associated with the suspended sediment does not change over time.
- *Yields of contaminants are correlated with urbanization.* Yields of metals, organochlorine compounds, and PAHs increased with an increase in developed land use. Yields of metals were most closely correlated with industrial land use, yields of organochlorine compounds were most closely correlated with either total developed or industrial land use, and yields of PAHs were most closely correlated with total developed land use.
- *Occurrence of hydrophobic contaminants in the urban creeks sampled are consistent with those found in Town Lake bed sediment.* Chlordane, DDT, and PAHs are among the contaminants that occur consistently in Town Lake bed sediment and were repeatedly detected in suspended sediment from Barton Above, Shoal, and Boggy. The active transport of these hydrophobic contaminants in urban creeks indicates that they continue

to be delivered to Town Lake, even though use of both DDT and chlordane has been banned.

- *Sediment discharging from Barton Springs is relatively uncontaminated.* Suspended sediment from Barton Springs had a median C_I of 0.12. Sediment quality in Barton Springs pool is more likely to be adversely affected by sediment from Barton Creek when the creek overtops the dam at the upstream end of the pool than by sediment discharging from the spring; the median C_I for suspended sediment from Barton Above was 0.34. Elevated concentrations of As, Cr, Cu, and Ni associated with suspended sediment discharging from Barton Springs appear to reflect the natural sediment geochemistry rather than an anthropogenic source. Concentrations of all metals at Barton Springs were less than the PECs.
- *Metals associated with suspended sediment make up the major part of the total storm load of metals.* Analysis of both dissolved- and particulate-phase metals for three storms at Shoal demonstrated that most of the metals loads in transport are associated with the particulate phase rather than the dissolved phase. Particle-associated loads of metals computed using LVSS data thus can provide estimates of total loads.

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Appendix 1—Evaluation of the Large-Volume Suspended-Sediment Approach in Relation to Traditional Sampling Approaches

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Large-volume suspended-sediment (LVSS) sampling is a departure from the two approaches traditionally followed for the evaluation of particle-associated (hydrophobic) contaminants in urban streams: the analysis of streambed sediment and the analysis of whole-water samples. LVSS sampling combines the advantages of both approaches without many of the disadvantages, as described below. It does, however, present its own unique difficulties, which are addressed at the end of this section.

Streambed-Sediment Sampling

Because hydrophobic contaminants partition to and are concentrated in sediment, analysis of streambed sediment, like LVSS, is more likely to detect the contaminants than water-column sampling. However, streambed sampling has limitations for characterization of the occurrence and transport of hydrophobic contaminants. First, streambed-sediment data cannot be used for computation of event loads or yields, because they are not associated with a known mass of sediment transported by the stream, whereas LVSS data can be combined with streamflow and suspended-sediment concentration (SSC) to compute event loads and yields. Second, streambed-sediment data appear to exhibit greater spatial and temporal variability than LVSS data in grain size and organic carbon content, two variables that affect contaminant concentrations. For example, on the basis of seven LVSS samples for which organic carbon was reported and nine streambed sediment samples collected at Barton Above (Leila Gosselink, City of Austin, unpub. data, 2005), the coefficient of variation (C_V) for organic carbon in the LVSS samples was 0.10, about 4 times less than the C_V of 0.38 for the streambed sediment. This variability is one of the reasons the U.S. Geological Survey (USGS) National Water Quality Assessment program has dropped streambed-sediment sampling as a means of evaluating relations between land use and water quality (R.J. Gilliom, U.S. Geological Survey, written commun., 2001). Further, streambed sediment varies spatially because of streambed topography, which influences where sediment is deposited, but suspended sediment in small, turbulent streams is mixed across the stream (John Gray, U.S. Geological Survey, oral commun., 2001). And third, streambed-sediment samples collected using USGS protocols (Shelton and Capel, 1994) theoretically represent a short period of time during an event—the time when streamflow decreases to the point that suspended sediment settles out of flow—whereas LVSS samples, if collected over the hydrograph and combined into a flow-weighted composite, integrate transport from the entire watershed and full event duration and represent an event-mean concentration.

Whole-Water Sampling

LVSS data offer advantages over whole-water data for characterization of the relative amount of sediment contamination and event-related transport of hydrophobic contaminants. Whole-water data can be used, in theory, for computation of event loads and yields of these contaminants, but often this is not possible because concentrations of hydrophobic contaminants are frequently at or below the minimum reporting level (MRL). For example, in 182 whole-water stormwater samples collected from 30 residential, commercial, and industrial sites in Fort Worth, Tex., chlordane was detected in less than 25 percent of samples, and benzo[*a*]pyrene was detected in less than 5 percent of samples (Baldys and others, 1998). Yet in 45 LVSS samples collected from 11 sites with a range of land uses in the same area, chlordane and benzo[*a*]pyrene were detected in more than 70 percent of samples (Van Metre and others, 2003). The frequency of detection is greater because hydrophobic contaminants are associated with sediment at concentrations many orders of magnitude greater than those in the dissolved (aqueous) phase. If the amount of sediment in the whole-water sample is small, the contaminant probably will not be detected in a laboratory analysis of whole water even if the sediment itself is highly contaminated.

A similar check can be done for all contaminants at other sites by converting the contaminant concentration in the LVSS samples to an equivalent whole-water concentration and comparing the result to laboratory MRLs. The conversion is carried out by multiplying the LVSS concentration (milligrams per kilogram for metals and micrograms per kilogram for organic compounds) by the SSC (milligrams per liter) and multiplying by a unit conversion factor of 10^{-3} for metals and 10^{-6} for organic compounds to obtain concentrations in micrograms per liter (table 1.1). In this conversion, the concentration of the contaminant in the dissolved phase is assumed to be negligible. The equivalent whole-water concentration determined from the LVSS sample can be compared to the USGS National Water Quality Laboratory (Denver, Colo.) MRL for the analytical schedule used for the whole-water analyses of metals and organochlorines performed, and for polycyclic aromatic hydrocarbons (PAHs) to the lowest MRL offered by the National Water Quality Laboratory. The LVSS detections and estimated whole-water detections also can be compared to measured analyses in whole water (fig. 1.1). Intermittently during 1978–96 the USGS collected whole-water samples, in cooperation with the City of Austin, for the analysis of selected organochlorine compounds in Shoal Creek (Buckner and others, 1982, 1993; Gandara and others, 1994–97; U.S. Geological Survey, 1979–81).

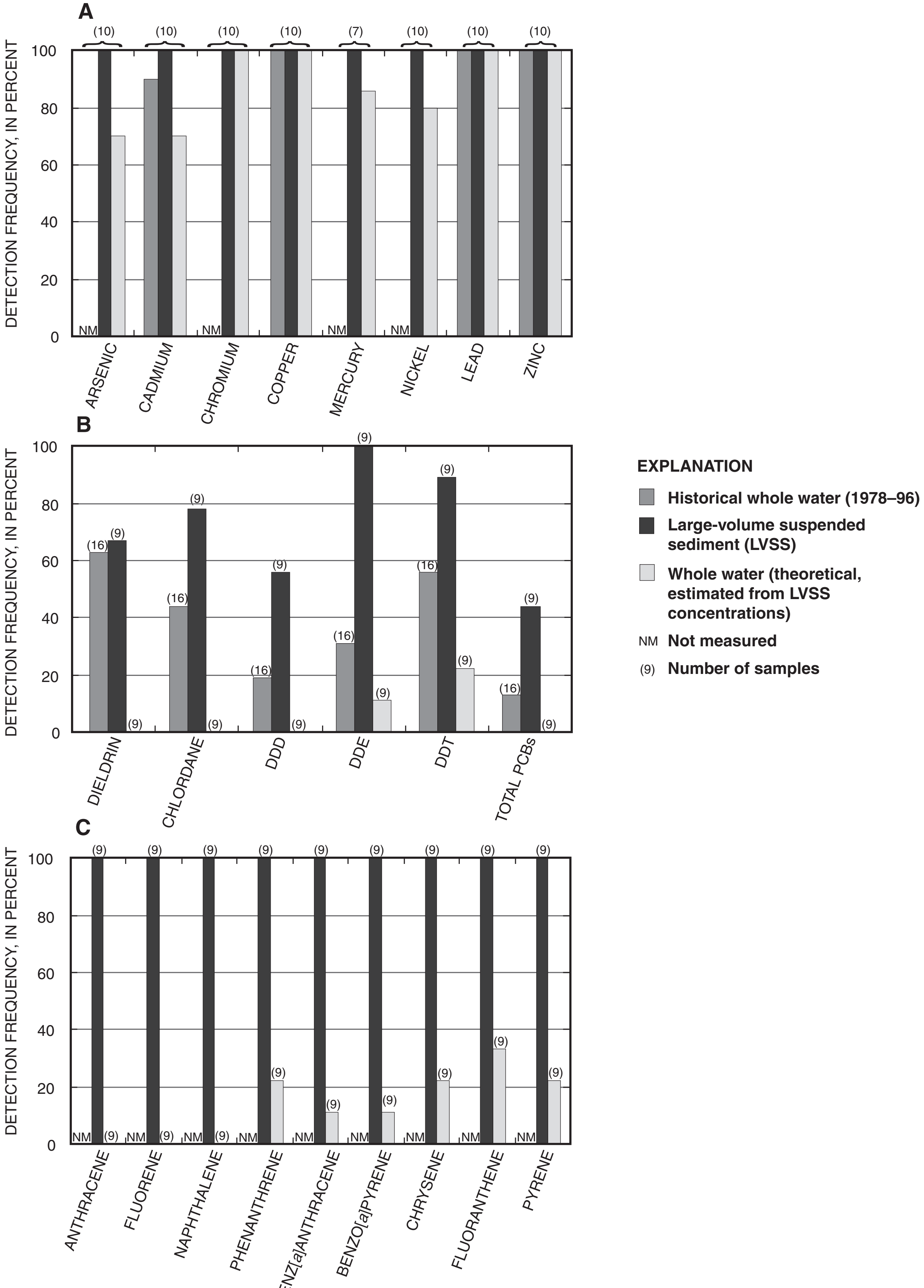

Figure 1.1. Comparison of detection frequency of (A) metals, (B) organochlorine compounds, and (C) polycyclic aromatic hydrocarbons (PAHs) at Shoal Creek in whole-water samples and LVSS samples, Austin, Texas, 1999–2004.

Table 1.1. Estimated concentrations of contaminants in whole water, computed on the basis of detected concentrations associated with suspended sediment.

[In micrograms per liter. Concentrations that would have been detected using standard USGS National Water Quality Laboratory whole-water methods in bold. est, estimated; NA, not analyzed; ND, not detected in large-volume suspended-sediment sample]

Date	Time	Arsenic (est)	Cadmium (est)	Chromium (est)	Copper (est)	Mercury (est)	Nickel (est)	Lead (est)	Zinc (est)
03/12/1999	1200	1.1	**0.26**	**5.8**	**5.7**	**0.017**	**2.6**	**16**	**50**
03/18/1999	2120–0144	**14.8**	**.82**	**84.6**	**29.8**	**.063**	**33.8**	**50**	**222**
03/28/1999	0045	**11.2**	.19	**65.0**	**16.3**	**.036**	**30.8**	**23**	**96**
04/26/1999	1347	**4.6**	**.22**	**29.7**	**11.8**	**.025**	**10.7**	**19**	**75**
10/18/1999	1520–1620	.4	.10	**2.1**	**2.2**	NA	.9	**5.5**	**15**
10/30/1999	1030–1510	**11.6**	**.58**	**90.4**	**27.3**	NA	**31.9**	**44**	**180**
01/07/2000	1700	**8.8**	**.40**	**61.5**	**20.6**	**.045**	**32.6**	**32**	**126**
03/17/2000	0410–1150	**2.3**	.13	**19.3**	**6.2**	.009	**6.9**	**8**	**43**
10/15/2000	1945	**13.7**	**1.0**	**83.5**	**28.4**	**.086**	**33.6**	**52**	**169**
04/23/2001	1300	.5	.05	**2.6**	**2.4**	NA	1.0	**3.9**	**14**
Minimum reporting level		1.9	.22	2.0	1.2	.01	2.0	1.0	10.0

Date	Time	Dieldrin (est)	Chlordane (est)	DDD (est)	DDE (est)	DDT (est)	Total PCBs (est)
03/12/1999	1200	ND	0.003	0.00066	0.00086	0.00063	ND
03/18/1999	2120–0144	.0013	.010	.0031	.0078	.0051	ND
04/26/1999	1347	.0013	.0044	ND	.0031	.0021	ND
10/18/1999	1520–1620	ND	.0077	ND	.0018	.0022	.0052
10/30/1999	1030–1510	.0013	ND	ND	.0033	.0027	ND
01/07/2000	1700	.0018	.020	.0015	.0048	**.014**	.030
03/17/2000	0410–1150	.0007	.011	.0003	.0019	.0039	.0025
10/15/2000	1945	.0072	.065	.0097	**.027**	**.029**	.026
04/23/2001	1300	ND	ND	ND	.0021	ND	ND
Minimum reporting level		.008	.10	.016	.014	.01	.10

Date	Time	Anthracene	Fluorene	Naphthalene	Phenanthrene	Benz[*a*]-anthracene	Benzo[*a*]-pyrene	Chrysene	Fluoranthene	Pyrene
03/12/1999	1200	0.013	0.005	0.006	0.077	0.058	0.052	0.15	0.22	0.14
03/18/1999	2120–0144	.11	.042	.018	**1.1**	.82	1.1	**1.8**	**2.4**	**1.9**
04/26/1999	1347	.024	.008	.007	.26	.18	.27	.49	.78	.62
10/18/1999	1520–1620	.011	.003	.004	.064	.073	.095	.16	.22	.18
10/30/1999	1030–1510	.044	.015	.015	.42	.27	.48	.77	1.3	1.0
01/07/2000	1700	.046	.021	.013	.55	.49	.74	.99	**1.7**	1.3
03/17/2000	0410–1150	.021	.006	.004	.20	.13	.20	.33	.49	.37
10/15/2000	1945	.17	.071	.027	**1.7**	**2.0**	**3.1**	**4.6**	**5.7**	**4.6**
04/23/2001	1300	.034	.012	.006	.14	.14	.27	.33	.52	.37
Minimum reporting level		2.0	1.2	1.6	1.0	1.6	1.3	1.2	1.4	1.6

Comparison of detection frequencies for the three groups of contaminants (metals, organochlorine compounds, and PAHs; fig. 1.1) shows that, at Shoal, metals in whole-water samples can be detected most successfully and PAHs least successfully. As presented in appendix 3, all metals were detected in all samples using the LVSS approach; three of the four metals analyzed in whole-water samples were detected in all of the samples. LVSS was more successful than whole-water analyses at detecting organochlorine compounds, particularly DDD, DDE, and PCBs. The difference between the two methods is most striking for PAHs, although only estimated concentrations in whole water can be compared. Individual PAHs were detected in every LVSS sample, and the estimated percentage that would have been detected using whole water ranged from 0 to about 35 percent.

Whole-water analyses of metals also were performed on samples from other sites in Austin, as presented in appendix 2. At these less contaminated sites, although metals consistently were detected in LVSS samples, there were fewer detections of metals in the whole-water samples. For example, at Barton Springs, cadmium was not detected in any samples, and lead was detected in only 30 percent of samples. Even at Barton Above, which is a more urban site, cadmium was detected in only 37 percent of samples.

The large number of nondetections of organochlorine compounds and PAHs gives the misleading impression that during many events these contaminants are not transported down Shoal Creek, but the number of detections in suspended-sediment samples demonstrates otherwise. Furthermore, when organochlorine compounds were detected in whole-water samples, in most cases the concentrations were near or at the MRL, which limits their precision and precludes their use for computing loads.

It is unlikely that changes in sampling, analytical protocol, or land use are the reason that organochlorine compounds historically were not detected in historical whole-water samples yet more recently are detected in suspended sediment. Sampling methods, analytical protocols, and MRLs for these samples have not changed greatly in the 10 to 35 years since the whole-water samples were collected. Land-use changes in the Shoal Creek watershed since the 1970s would be expected to have a minimal effect, because use of the organochlorine compounds in question is no longer permitted. Furthermore, because use of these compounds was banned in the 1970s and 1980s, whole-water samples collected 15 to 25 years ago would have been more likely to contain detectable concentrations of these contaminants than if those samples were collected now. This might explain why the frequency of detection of organochlorine compounds in historical whole-water samples collected during 1978–96 exceeds the estimated frequency of detections in samples collected during 1999–2004 (fig. 1.1). Therefore the most likely reason for the discrepancy in detection of organochlorine compounds between the whole-water samples and suspended-sediment samples remains the medium: LVSS is a better tool for determining the occurrence of hydrophobic compounds than whole-water analysis.

Limitations of LVSS

Although LVSS has been shown to have advantages over streambed-sediment and whole-water sampling, it presents some drawbacks. First, there are no regulatory limits or sediment-quality guidelines (SQGs) associated with suspended sediment. There are no SQGs, because SQGs are based on toxicity of sediment to benthic organisms; although some studies demonstrate the toxic effects of suspended sediment contaminated with hydrophobic compounds on pelagic organisms (Bonnet and others, 2000; Swartz and others, 2003), this research is in its infancy. This could be problematic for governments or agencies trying to determine how to respond to contaminated suspended sediment. Until such time as regulations are promulgated for suspended sediment, LVSS is better suited as an investigative tool than as a regulatory tool.

Second, when using automated samplers with preset volumes, the amount of sediment obtained might be insufficient for analysis. This can occur in streams that typically have low SSC, or if recent rains have removed most of the easily eroded street dust and soil. Sample volume rarely is a problem with either whole-water or streambed-sediment samples, although low concentrations of suspended solids often result in nondetection of the target analytes using whole-water methods, as discussed above.

Third, because the laboratory reporting level (LRL) depends to some degree on the amount of sediment available for analysis and the associated mass of contaminant, LRLs for the same constituent can vary from one sample to the next. This can complicate statistical analysis of the data because techniques appropriate for multiply-censored data, such as survival analysis, must be used.

Fourth, collection and filtration of suspended sediment for analysis is more labor-intensive than collection of streambed-sediment samples or whole-water samples. The filtration process used here requires specialized equipment and supplies, although the cost of the equipment and supplies is less than for other sediment separation techniques such as centrifugation.

Appendix 2—Total Concentrations of Metals in Whole Water and Comparison to Concentrations in Large-Volume Suspended-Sediment Samples

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During the period of study, the U.S. Geological Survey (USGS), in cooperation with the City of Austin, analyzed metal concentrations in whole (unfiltered) stormwater samples from the same sites as those for the study presented in this report, although not necessarily for the same events (Aragon Long and others, 2005; Gandara, 2002–04; Gandara and others, 2000, 2001). Samples for analysis of total cadmium (Cd), copper (Cu), lead (Pb), and zinc (Zn) were collected for three storms each year. Total analyses measure the sum of the metals in the dissolved and particulate phases without discriminating between the two. If most of the mass of a metal in a whole-water sample is associated with the particulate matter, as hypothesized, then the total metal concentration will increase with the suspended-solids concentration. The relation between the two will define an average metal concentration associated with the particulate phase on a mass per mass basis, similar to what is determined directly by large-volume suspended-sediment (LVSS) sampling.

Samples for analysis of metals in whole water were collected and combined into flow-weighted composites using automated ISCO samplers and either 1-liter (L) plastic bottles or 8-L polytetrafluoroethylene bags housed in 9-L carboys. The flow-weighted composite sample was mixed in a churn splitter, and the whole-water sample for analysis was obtained while churning at a rate of about 9 inches per second. Each sample bottle was rinsed and then filled from a spigot located near the bottom of the churn. Total metals samples were preserved using an amount of concentrated nitric acid sufficient to lower the pH to less than 2.

The samples were analyzed at the USGS National Water Quality Laboratory, Denver, Colo. To determine the suspended-solids concentration (reported by this method as "insoluble residue" and analogous to suspended-sediment concentration) in the whole-water sample, a split of the unfiltered sample is mixed thoroughly and an appropriate volume is poured rapidly into a graduated cylinder. The insoluble residue is collected on a glass-fiber filter, dried at 105 degrees Celsius, and weighed. For metals analysis, a split of the unfiltered sample is treated with concentrated hydrochloric acid and digested in a capped bottle for 8 hours (Hoffman and others, 1996). This is a less aggressive digestion than that used for the LVSS samples, and in most cases not all solids are digested (J.R. Garbarino, U.S. Geological Survey, oral commun., 2005). Any remaining solids are filtered out, and the filtrate is analyzed by graphite furnace (Cd, Cu, and Pb) or inductively-coupled plasma-mass spectrometry (Zn) (Fishman, 1993; Garbarino and Struzeski, 1998).

Total concentrations of Cd, Cu, Pb, and Zn for Barton 71, Barton Above, Barton Springs, Shoal, Boggy, and Williamson are listed in table 2.1. Concentrations cannot be compared to Texas Commission on Environmental Quality criteria for aquatic life protections or to U.S. Environmental Protection Agency maximum contaminant levels for drinking water, as those criteria apply to dissolved rather than total concentrations. At three sites (Barton 71, Barton Above, and Barton Springs), one-half or more of all Cd concentrations were below the minimum reporting level (MRL), and at Williamson they were either below or very near the MRL. On the basis of median concentrations, the highest Cd, Cu, and Pb concentrations were detected at Boggy Creek, and the highest Zn concentrations were detected at Shoal Creek.

Table 2.1. Metals concentrations in whole-water samples.

[µg/L, micrograms per liter; mg/L, milligrams per liter; <, less than; E, estimated, --, not analyzed]

Site	Date	Time	Cadmium, filtered (µg/L)	Copper, filtered (µg/L)	Lead, filtered (µg/L)	Zinc, filtered (µg/L)	Residue, unfiltered (mg/L)
Barton 71	05/26/1999	2030	<1	2	3	<40	77
Barton 71	06/20/1999	1755	<1	<1	<1	<40	8
Barton 71	07/11/1999	0555	<1	<1	<1	<40	<1
Barton 71	05/02/2000	0005	<.1	E.9	--	<31	35
Barton 71	06/09/2000	0930	.1	3.3	5	<31	338
Barton 71	11/02/2000	2030	.2	9.2	12	35	780
Barton 71	05/06/2001	2320	E.1	4.1	6	21	221
Barton 71	11/15/2001	1155	.2	9.5	14	35	1,120
Barton 71	06/30/2002	0505	E.1	6.6	8	20	576
Barton 71	07/16/2002	0855	.8	3	4	22	<40
Barton 71	10/19/2002	1019	<.2	E1.2	2	10	72
Barton 71	12/09/2002	0510	<.2	<1	<1	3	17
Barton 71	02/20/2003	0920	E.1	2.9	6	16	464
Barton 71	02/29/2004	1105	E.03	3	1.8	8	101
Barton 71	04/06/2004	1525	<.04	1.6	.5	4	21
Barton Above	05/17/1999	2215	<1	5.3	16	E30	250
Barton Above	05/26/1999	1830	<1	3	7	E25	121
Barton Above	06/21/1999	1610	<1	1.4	<1	<40	8
Barton Above	07/10/1999	2120	<1	3.9	8	E29	130
Barton Above	05/02/2000	0135	.2	2.9	4	E21	25

Table 2.1. Metals concentrations in whole-water samples—Continued.

Site	Date	Time	Cadmium, filtered (μg/L)	Copper, filtered (μg/L)	Lead, filtered (μg/L)	Zinc, filtered (μg/L)	Residue, unfiltered (mg/L)
Barton Above	06/09/2000	2050	.4	1.5	1	E19	33
Barton Above	11/02/2000	1940	E.1	2.4	3	37	58
Barton Above	03/12/2001	0130	E.1	3.1	9	27	134
Barton Above	05/06/2001	2135	<.1	3.6	10	30	150
Barton Above	11/15/2001	1540	.2	6.5	16	44	248
Barton Above	06/30/2002	0455	<.1	3.6	4	13	282
Barton Above	10/19/2002	1210	<.2	E1.2	1	8	<10
Barton Above	12/09/2002	0315	<.2	<1	<1	2	<10
Barton Above	02/06/2003	0650	<.2	<1	<1	4	<10
Barton Above	01/17/2004	0615	E.03	1.9	.69	8	15
Barton Above	04/07/2004	0115	<.12	E1.2	.36	E4	<10
Barton Springs	05/27/1999	1150	<1	<1	<1	<40	2
Barton Springs	06/22/1999	0930	<1	<1	<1	<40	2
Barton Springs	06/10/2000	0845	<.1	1.4		<31	19
Barton Springs	05/07/2001	1430	<.1	<1.2	<1	1	<10
Barton Springs	11/16/2001	1200	<.1	4.6	4	10	152
Barton Springs	06/30/2002	1600	<.04	1.1	<1	2	--
Barton Springs	11/06/2002	1243	<.2	<1	<1	<2	<10
Barton Springs	12/10/2002	1030	<.2	<1	<1	<2	<10
Barton Springs	02/20/2003	1845	<.2	<1	<1	<2	<10
Barton Springs	01/17/2004	0830	<.04	1.4	.39	E1	<10
Barton Springs	06/09/2004	0900	<.04	.9	.1	<2	<10
Shoal Creek	03/28/1999	0048	<1	12.9	18.5	66	792
Shoal Creek	04/26/1999	0910	--	--	--	--	--
Shoal Creek	10/18/1999	1515	.1	10.2	11	46	38
Shoal Creek	10/30/1999	1131	.3	16.8	25.8	116	1,000
Shoal Creek	12/12/1999	0553	.2	10	15.4	53	254
Shoal Creek	10/15/2000	1950	.4	15.9	32.2	111	960
Shoal Creek	03/27/2001	1035	.2	9.6	21.9	63	<10
Shoal Creek	10/11/2001	0510	.3	30	60.4	129	294
Shoal Creek	06/30/2002	0125	.2	10.4	18.8	216	508
Shoal Creek	10/19/2002	0545	.3	13.6	19.1	138	672
Shoal Creek	12/23/2002	0730	.6	31.4	55.6	164	1,820
Boggy Creek	03/18/1999	2130	<1	23.2	59	148	1,120
Boggy Creek	05/10/1999	0420	<1	18.7	30.9	99	576
Boggy Creek	10/30/1999	1231	.3	12.7	19.5	68	623
Boggy Creek	01/07/2000	1300	.3	12.1	23.8	81	523
Boggy Creek	02/23/2000	0005	.7	19.2	48.9	143	875
Boggy Creek	10/15/2000	2105	.5	14.9	47.7	108	940
Boggy Creek	01/10/2001	0940	.8	9	20	60	344
Williamson Creek	03/19/2002	2115	E.1	5.3	11	49	204
Williamson Creek	06/16/2002	0455	<.1	2.9	3	23	63
Williamson Creek	10/08/2002	1250	<.2	3	3	18	33
Williamson Creek	02/20/2003	0415	E.1	4.5	9	30	190
Williamson Creek	11/17/2003	1600	.34	14.3	29.5	118	446
Williamson Creek	04/29/2004	0240	E.03	1.9	3.4	14	44

Of interest is whether total metal concentrations in water vary linearly with the insoluble residue (suspended solids). If so, the mass of the metal in the water is associated mostly with the suspended solids, and the suspended solids have a relatively unvarying metals concentration. If this is the case and the relation is established, the metal concentration can be predicted if the suspended-solids concentration is known.

Each total metal concentration at each site was regressed against insoluble residue concentration to determine if there was a linear relation (table 2.2). Of the 24 possible cases (four metals at six sites), seven could not be tested because of the frequency of nondetections. Of the remaining 17 cases, there was a statistically significant linear relation (p-value < .05) for 14 cases (table 2.2). In two of those cases (Pb at Barton Above on 6/30/2002 and Pb at Shoal on 10/11/2001) a single outlier was removed to determine the relation. In 13 of the 14 cases, the linear regression coefficient (r^2) was .65 or greater, which indicates that the majority of the variability in the concentrations is accounted for by the suspended solids. In other words, for more than 80 percent of the analyses that were run, the total metal concentration is mostly a measure of the suspended-solids concentration.

Table 2.2. Linear regression coefficient for relation between individual total metal concentrations and insoluble residue (suspended-solids) concentrations.

[Value in italics obtained by removing a single outlier. r^2, linear regression coefficient; p-value, associated significance level; m, slope of regression; LVSS, large-volume suspended sediment; mg/L, milligrams per liter; --, not detected; *, regression not significant (p-value $\geq$ .05)]

Site	Cadmium				Copper			
	r^2	p-value	m (X 1,000)	Mean concentration, LVSS samples (mg/L)	r^2	p-value	m (X 1,000)	Mean concentration, LVSS samples (mg/L)
Barton 71	--	--	--	--	.90	0	8.2	15.0
Barton Above	--	--	--	--	.77	0	14.5	21.7
Barton Springs	--	--	--	--	--	--	--	--
Shoal	.85	0	0.2	1	*	.05	*	*
Boggy	*	.05	*	*	.65	.03	14.6	22.4
Williamson	--	--	--	--	.93	0	28.1	28.2

Site	Lead				Zinc			
	r^2	p-value	m (X 1,000)	Mean concentration, LVSS samples (mg/L)	r^2	p-value	m (X 1,000)	Mean concentration, LVSS samples (mg/L)
Barton 71	.96	0	12.4	19.3	.86	0	28.7	94.5
Barton Above	*.98*	*0*	62.9	43.4	.36	.02	80.4	187
Barton Springs	--	--	--	--	--	--	--	--
Shoal	*.77*	*0*	20.5	64.3	*	.05	*	*
Boggy	.88	0	55.4	40.9	.78	.01	112	148
Williamson	.97	0	64.2	58.3	.94	0	241	276

In those cases where the linear relation was significant, the accompanying linear equation (in the form $y = mx + b$) was determined, where x is the insoluble residue concentration (milligrams per liter), and y is the total metal concentration (micrograms per liter). In this case, the slope m represents the mean concentration of the metal associated with the suspended solids (micrograms of metal per milligram of residue); a conversion factor of 1,000 is used to convert the concentration to milligrams of metal per kilogram of solid, the same units used for the LVSS samples.

When the mean metals concentrations associated with suspended solids in the whole-water samples are compared among sites, the results are somewhat different than for the total metal concentrations: The highest mean Cu and Zn concentrations are at Williamson Creek (table 2.2), not at Boggy and Shoal Creeks, respectively, as they were in whole water (table 2.1). Cu and Zn are the two metals most closely associated with vehicles—Cu from brake wear (U.S. Environmental Protection Agency, 2001) and Zn from tire wear (Councell and others, 2004)—which indicates that road and parking lot runoff might be an important source of metals in the Williamson Creek watershed. This is consistent with the high polycyclic aromatic hydrocarbons (PAHs) concentrations in Williamson Creek, which also might be associated with parking lot runoff, as discussed in the section on PAHs under "Sample Concentrations, Loads, and Yields." The site with the highest Cd concentrations associated with suspended solids cannot be determined because there was no relation between suspended solids and Cd at Boggy Creek (fig. 2.1), one of only two sites where there were enough detections to test. Therefore, although the highest total metal concentrations in whole water were at Boggy Creek because it carries the highest load of suspended solids, the solids themselves are more contaminated at Shoal and Williamson Creeks. This same conclusion was reached by comparing metals concentrations in the suspended sediment measured using LVSS.

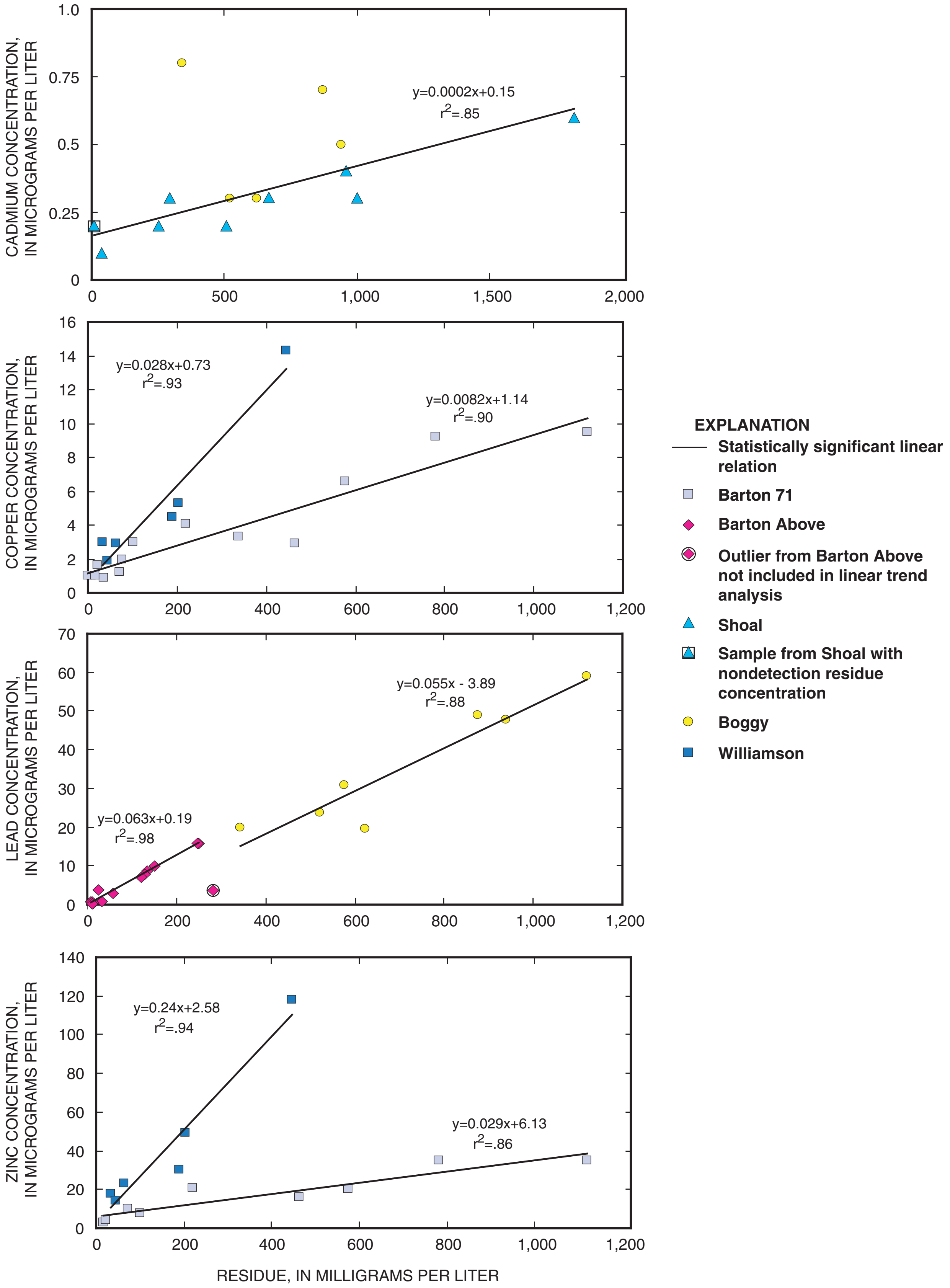

Figure 2.1. Examples of linear relations between total metal concentration in whole water and suspended-solids concentration in the samples in urban creeks, Austin, Texas, 1999–2004.

Each metal concentration at each site associated with the suspended solids estimated from the total metal concentrations was compared to the mean metal concentration in the LVSS samples (table 2.2). In most cases the concentration determined for the LVSS samples is higher than that associated with suspended solids in the whole-water samples. This was most pronounced at Barton 71, which is the least urbanized and least contaminated of the surface-water sites. Concentrations were more similar at the four other sites, particularly Williamson. This is because the digestion for the whole-water samples is less aggressive than that for the sediment samples, and because not all the solids are digested. At sites where most of the metals concentration is nonanthropogenic, that is, it is bound up in the mineral matter, the digestion used for the total metals analysis would be expected to recover less of the metal than the complete digestion used for the LVSS analysis. If, however, most of the metal associated with the sediment has an anthropogenic origin and is bound to the surface of the particle, it will be equally efficiently extracted using either digestion, and the mineral component will be less important.

Analysis of total metal concentrations in whole water has advantages and disadvantages compared to LVSS. The major advantage is that the sample is easier to process, as no filtration is required. A disadvantage is that concentrations determined by this method reflect the amount of suspended solids in the water rather than the actual contamination of the solids—a higher suspended-sediment (insoluble residue) concentration translates into a higher total metal concentration, regardless of the degree of contamination of the sediment. Thus by total metals analysis, Boggy Creek has more contaminated water because it carries more suspended sediment, but the sediment itself is less contaminated than at other sites. Second, although total metals analyses can be used to determine the concentration associated with the sediment, as shown, the method requires analysis of a sufficient number of samples to establish the linear relation, whereas the concentration associated with the sediment can be determined with a single LVSS sample. Third, if suspended-sediment concentration is low, analysis of whole water often results in a nondetection. This happened numerous times for Cd; at Barton Springs more than 75 percent of all samples resulted in nondetections, which yielded very little useful information. In contrast, there were no nondetections of these four metals at any of the sites using the LVSS sampling approach.

An argument could be made that total metals analysis provides information on the contaminant concentration in the water column to which the biota are subjected, but this might not accurately reflect biological processes. For example, at the Donna Canal, Hidalgo County, Tex., fish have highly elevated concentrations of polychlorinated biphenyls (PCBs) yet no PCBs have been detected in whole water or bed sediment (Mahler and Van Metre, 2003; Mahler and others, 2002). The only medium in which PCBs have been detected in Donna Canal is suspended sediment. Thus results of whole-water analyses might give an erroneous view of the contamination to which biota are exposed.

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Appendix 3—Concentrations Associated With Suspended Sediment in Composite Rain Event Samples

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Appendix 3.1. Suspended-sediment concentrations and organic carbon, major element, and metals concentrations associated with suspended sediment in composite rain event samples.

[In milligrams per kilogram except where noted; mean value listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
SSC, suspended-sediment concentration; mg/L, milligrams per liter; --, not analyzed or not applicable; na, not analyzed because of insufficient sediment mass; C_v, coefficient of variation; TEC, threshold effect concentration (MacDonald and others, 2000); PEC, probable effect concentration (MacDonald and others, 2000)]

Site	Date	Time (24-hour)	SSC (mg/L)	Sand content (percent)	Organic carbon (percent)	Aluminum	Calcium	Iron	Potassium	Magnesium
Barton 71	05/26/1999	2300	64.7	--	na	70,100	133,000	24,500	16,400	7,710
Barton 71	05/01/2000	2255	41	--	na	35,400	213,000	17,400	9,580	14,300
Barton 71	[1]06/09/2000	0930	476.5	1.9	2.86	42,400	219,000	20,300	11,200	10,800
Barton 71	11/02–03/2000	1900	899.5	.1	2.27	50,700	176,000	22,100	12,300	11,600
Barton 71	05/06/2001	2320	589	--	2.76	37,900	189,000	19,900	9,680	9,070
Barton 71	11/15/2001	0945	1,183	2.2	4.23	35,200	217,000	16,900	8,800	12,800
Barton 71	[1]06/30/2002	0505–1735	660	1.0	2.60	42,200	195,000	19,700	9,790	13,700
Barton 71	[1]02/20/2003	0920–0430	484	3.8	2.81	35,800	207,000	17,000	8,440	9,540
Barton 71	06/04/2003	0220	279	.9	2.13	40,700	198,000	19,500	9,660	9,800
Barton 71	02/29/2004	1105	114	1.1	3.42	39,800	213,000	19,000	10,100	9,760
Mean			479	1.6	2.89	43,020	196,000	19,630	10,595	10,908
C_v			.78	.76	.23	.25	.13	.12	.22	.20
Barton Above	05/18/1999	0100	264	--	3.28	64,200	154,000	25,400	13,800	11,400
Barton Above	05/26/1999	2100	116.4	4.0	3.60	65,400	144,000	25,300	14,100	12,000
Barton Above	05/02/2000	0135	28	--	na	64,500	96,000	29,800	14,900	13,200
Barton Above	06/09/2000	2055	105.5	2.1	3.89	62,900	239,000	29,400	15,400	15,500
Barton Above	11/03/2000	1940–0520	48	1.3	na	66,600	108,000	28,300	13,400	14,100
Barton Above	05/06/2001	2135	146	--	na	38,800	144,000	19,400	8,370	10,100
Barton Above	08/26/2001	2235	159	.5	4.28	65,100	134,000	27,500	11,600	11,400
Barton Above	[1]11/15/2001	1535	670	9.5	3.34	50,900	202,000	21,700	10,300	13,600
Barton Above	06/30/2002	0455–2355	227	.1	3.64	44,700	188,000	20,500	10,200	12,400
Barton Above	02/20/2003	0510–0945	202	3.0	3.40	34,800	206,000	16,200	7,860	8,790
Barton Above	01/17/2004	0600	14	0	na	na	na	na	na	na
Mean			180	2.6	3.63	55,790	161,500	24,350	11,993	12,249
C_v			1.0	1.2	.10	.18	.29	.15	.20	.13
Barton Springs	[1]05/27/1999	1440	8.875	16	2.93	82,000	59,100	36,900	12,800	24,800
Barton Springs	05/01/2000	1200	3.6	--	2.27	88,800	71,300	42,200	13,300	33,500
Barton Springs	05/02/2000	0200	7.0	0	na	72,600	85,400	35,700	11,800	39,400
Barton Springs	06/10/2000	1020	40.0	--	1.58	62,300	84,600	31,300	10,300	40,500
Barton Springs	11/03/2000	1030	44	2.9	2.09	69,500	74,500	34,400	10,600	41,600
Barton Springs	05/07/2001	0940	47	47	1.67	34,100	130,000	18,400	6,020	51,400
Barton Springs	08/27/2001	1314	12	0	na	70,500	74,400	33,100	10,800	39,800
Barton Springs	11/16/2001	1130	90.75	0	3.56	88,600	32,800	42,600	11,400	17,900
Barton Springs	06/30/2002	1650–2330	11	0	2.07	62,300	81,400	28,900	9,070	51,800
Barton Springs	02/20–21/2003	2120–0105	9	0	na	70,600	53,000	35,300	12,100	22,100
Barton Springs	01/17/2004	0815	39.6	.5	1.20	44,900	129,000	22,000	6,810	72,600
Barton Springs	06/09/2004	0900	7.8	0	na	50,000	99,600	25,000	7,990	47,300
Mean			27	6.6	2.17	66,350	81,258	32,150	10,249	40,225
C_v			.98	2.3	.40	.25	.35	.23	.23	.37

Footnotes at end of table.

Appendix 3.1. Suspended-sediment concentrations and organic carbon, major element, and metals concentrations associated with suspended sediment in composite rain event samples—Continued.

Site	Date	Time (24-hour)	SSC (mg/L)	Sand content (percent)	Organic carbon (percent)	Aluminum	Calcium	Iron	Potassium	Magnesium
Shoal Creek	[2]03/12/1999	1200	83	0	7.49	48,800	162,000	25,400	12,700	10,600
Shoal Creek	03/18/1999	2120–0144	1,210	2.1	2.88	70,700	147,000	25,600	16,700	8,580
Shoal Creek	03/28/1999	0045	910	.4	1.53	85,800	95,600	28,400	18,900	8,140
Shoal Creek	04/26/1999	0910–1347	410.6	1.1	3.42	60,300	110,000	26,200	18,500	7,760
Shoal Creek	10/18/1999	1520–1620	30.6	--	na	47,000	59,500	23,400	13,000	9,540
Shoal Creek	10/30/1999	1030–1510	1,107	--	2.07	83,000	111,000	26,800	18,100	8,990
Shoal Creek	[1]01/07/2000	1700	901.5	3.2	2.58	66,700	112,000	24,400	16,200	7,720
Shoal Creek	03/17/2000	0410–1150	205	--	na	88,400	104,000	30,000	21,200	7,720
Shoal Creek	10/15/2000	1945	1,222	4.1	2.46	59,200	123,000	22,800	15,700	9,170
Shoal Creek	04/23/2001	1300	43	1	na	50,200	95,600	20,000	10,800	9,640
Mean			612	1.7	3.31	66,010	111,970	25,300	16,180	8,786
C_v			.82	.89	.65	.24	.25	.11	.20	.11
Boggy Creek	03/18/1999	2200	716	3.2	2.40	55,300	178,000	24,600	12,400	9,820
Boggy Creek	05/10/1999	0500	796	8.3	2.73	52,200	175,000	23,600	11,900	7,890
Boggy Creek	10/30/1999	1130	375	--	1.91	63,200	143,000	26,100	13,600	9,060
Boggy Creek	01/07/2000	1700	942.5	4.2	2.17	50,300	146,000	23,700	12,600	7,940
Boggy Creek	[1]02/23/2000	0100	1,333.5	5.0	na	59,800	173,000	21,500	14,900	8,030
Boggy Creek	10/15/2000	2105–0110	1,844	9.1	2.23	47,800	149,000	22,000	11,600	9,720
Boggy Creek	01/10/2001	0940–1400	414	4.5	4.35	47,800	137,000	22,900	10,900	8,750
Boggy Creek	[1]05/25/2001	0205	859	--	2.28	51,800	165,000	20,600	12,200	7,820
Mean			910	5.7	2.58	53,525	158,250	23,125	12,513	8,629
C_v			.53	.42	.47	.10	.10	.08	.10	.10
Williamson Creek	03/17/2000	0315	23	--	na	70,200	106,000	26,000	16,000	9,480
Williamson Creek	05/01/2000	0400	402	5.1	4.20	61,000	143,000	27,600	12,800	12,400
Williamson Creek	11/03/2000	1410–1535	316	6.4	3.87	50,200	119,000	23,100	10,600	13,000
Williamson Creek	05/26/2001	1245	118	--	4.32	67,400	103,000	24,500	14,000	9,790
Williamson Creek	[1]10/11/2001	0505	186	--	4.28	72,100	94,100	26,600	15,400	8,510
Williamson Creek	[1]03/19/2002	2200	179	.5	5.73	53,900	117,000	23,500	11,300	7,560
Williamson Creek	02/20/2003	0420–2015	195	33	4.65	59,400	104,000	25,600	11,400	8,520
Williamson Creek	03/25/2003	1950	113	2.3	6.10	66,400	132,000	25,100	14,700	9,850
Williamson Creek	11/17/2003	1555	590	9.9	5.03	60,400	107,000	25,400	13,100	8,130
Mean			236	9.5	4.77	62,333	113,900	25,267	13,256	9,693
C_v			.74	1.3	.17	.12	.14	.06	.14	.19
TEC			--	--	--	--	--	--	--	--
PEC			--	--	--	--	--	--	--	--

Footnotes at end of table.

Appendix 3.1. Suspended-sediment concentrations and organic carbon, major element, and metals concentrations associated with suspended sediment in composite rain event samples—Continued.

Site	Date	Sodium	Phosphorus	Titanium	Arsenic	Cadmium	Chromium	Copper	Mercury	Nickel	Lead	Zinc
Barton 71	05/26/1999	1,070	740	3,120	11.2	0.55	66.6	20.8	0.06	23.5	39.2	141
Barton 71	05/01/2000	3,930	682	1,700	6.8	1.9	32.6	18.6	na	30.2	28.9	271
Barton 71	[1]06/09/2000	887	523	2,050	8.2	.35	37.2	12.8	.03	21.3	15.7	57.6
Barton 71	11/02–03/2000	683	509	2,720	9.6	.40	46.9	16.1	.03	23.1	16.2	67.5
Barton 71	05/06/2001	561	436	2,150	7.3	.23	33.9	12.2	.04	21.1	13.3	67.8
Barton 71	11/15/2001	572	396	1,990	5.5	.26	30.7	11.6	.04	17.5	15.3	54.7
Barton 71	[1]06/30/2002	946	445	1,860	7.0	.38	36.3	16.1	.03	20.1	13.7	65.3
Barton 71	[1]02/20/2003	861	414	3,920	7.6	.23	28.4	10.3	.03	22.8	14.3	57.5
Barton 71	06/04/2003	151	466	4,530	8.0	.27	35.4	13.5	.03	24.6	17.2	62.8
Barton 71	02/29/2004	589	500	2,200	8.8	.38	41.8	17.8	na	10.7	19.3	100
Mean		1,025	511	2,624	8	.50	39	15	.04	21	19	95
C_v		1.03	.22	.36	.20	1.02	.28	.23	.28	.23	.43	.71
Barton Above	05/18/1999	778	1,010	2,830	12.3	.48	67.5	21.4	.07	24.3	56.2	168
Barton Above	05/26/1999	1,300	986	2,780	12.0	.55	64.0	20.4	.09	23.6	55.8	214
Barton Above	05/02/2000	2,170	1,960	2,630	15.9	3.5	69.0	31.1	.14	41.0	76.8	326
Barton Above	06/09/2000	1,900	1,060	2,830	11.7	.71	61.5	21.6	.04	37.9	35.8	301
Barton Above	11/03/2000	1,170	1,260	2,960	16.2	.77	67.4	24.7	.03	28.2	56.8	210
Barton Above	05/06/2001	845	1,090	2,240	9.3	.43	44.4	22.2	na	22.1	52.6	204
Barton Above	08/26/2001	710	1,010	3,500	14.5	.50	63.2	19.2	.04	25.5	33.0	160
Barton Above	[1]11/15/2001	1,390	804	2,710	11.6	.31	47.0	15.7	.06	20.6	32.7	106
Barton Above	06/30/2002	870	598	1,830	7.9	.32	40.8	18.1	.04	21.2	16.9	107
Barton Above	02/20/2003	891	452	3,890	7.0	.24	29.5	23.0	.04	22.2	17.2	77.4
Barton Above	01/17/2004	na	na	na	na	na	na	na	na	na	na	na
Mean		1,202	1,023	2,820	12	.78	55	22	.06	27	43	187
C_v		.42	.35	.17	.27	1.24	.25	.19	.57	.27	.45	.44
Barton Springs	[1]05/27/1999	4,840	996	3,700	20.0	.35	68.6	36.9	.09	34.1	25.2	(3)
Barton Springs	05/01/2000	3,050	884	4,000	22.3	.41	80.7	38.0	.08	40.1	31.7	113
Barton Springs	05/02/2000	4,870	815	3,330	19.3	.32	67.5	27.8	.09	34.6	24.6	120
Barton Springs	06/10/2000	1,010	757	3,000	17.8	.37	82.1	26.5	.07	40.5	21.3	87.8
Barton Springs	11/03/2000	804	884	4,280	21.0	.73	68.6	28.6	.07	32.0	22.4	91.3
Barton Springs	05/07/2001	549	672	1,970	10.5	.40	36.3	17.6	.08	24.2	12.6	50.4
Barton Springs	08/27/2001	784	826	3,440	18.5	.37	62.6	25.7	na	31.6	23.0	100
Barton Springs	11/16/2001	1,030	891	5,140	22.8	.58	82.8	40.8	.07	44.0	31.8	133
Barton Springs	06/30/2002	1,010	697	2,800	17.3	.30	61.9	28.0	.06	31.7	19.3	108
Barton Springs	02/20/2003	1,720	1,420	4,180	20.8	.50	66.6	33.5	na	42.6	29.0	132
Barton Springs	01/17/2004	810	470	2,100	11.0	.22	40.4	16.7	.04	22.2	14.1	61.6
Barton Springs	06/09/2004	713	580	2,600	14.0	.28	54.4	21.4	.05	27.4	17.8	72.7
Mean		1,766	824	3,378	18	.40	64	28	.07	34	23	97
C_v		.90	.29	.28	.23	.35	.23	.27	.23	.21	.27	.28

Footnotes at end of table.

Appendix 3.1. Suspended-sediment concentrations and organic carbon, major element, and metals concentrations associated with suspended sediment in composite rain event samples—Continued.

Site	Date	Sodium	Phosphorus	Titanium	Arsenic	Cadmium	Chromium	Copper	Mercury	Nickel	Lead	Zinc
Shoal Creek	[2]03/12/1999	2,020	1,950	2,510	13.7	3.2	69.6	69.2	0.20	30.8	196	607
Shoal Creek	03/18/1999	1,140	903	2,930	12.2	.68	70.0	24.7	.05	27.9	41.0	183
Shoal Creek	03/28/1999	939	692	3,250	12.3	.21	71.4	17.9	.04	33.9	25.4	105
Shoal Creek	04/26/1999	1,420	1,140	3,210	11.3	.54	72.3	28.8	.06	26.1	47.2	182
Shoal Creek	10/18/1999	2,200	5,890	2,540	11.9	3.2	68.0	72.9	na	30.8	178	504
Shoal Creek	10/30/1999	891	1,030	3,690	10.5	.53	81.6	24.6	na	28.8	39.6	163
Shoal Creek	[1]01/07/2000	929	711	3,420	9.8	.45	68.2	22.8	.05	36.2	36.0	140
Shoal Creek	03/17/2000	1,510	1,460	4,460	11.2	.65	94.3	30.1	.04	33.9	38.3	211
Shoal Creek	10/15/2000	1,010	819	2,910	11.2	.83	68.3	23.2	.07	27.5	42.9	138
Shoal Creek	04/23/2001	1,330	2,420	2,360	11.1	1.1	61.0	54.9	na	23.8	90.8	320
Mean		1,339	1,701	3,128	12	1.1	72	37	.07	30	74	255
C_v		.34	.93	.20	.09	.98	.13	.56	.78	.13	.85	.67
Boggy Creek	03/18/1999	1,750	812	2,540	10.1	.64	52.5	23.8	.05	23.9	50.3	168
Boggy Creek	05/10/1999	1,370	872	2,630	10.2	.54	51.4	23.2	.06	22.3	40.9	151
Boggy Creek	10/30/1999	2,090	928	3,100	9.7	.56	58.0	24.1	.04	26.7	33.0	136
Boggy Creek	01/07/2000	2,080	757	2,890	8.6	.72	50.9	21.3	.04	40.5	36.7	134
Boggy Creek	[1]02/23/2000	2,110	1,050	3,770	10.8	.69	69.4	24.5	.06	30.8	35.7	160
Boggy Creek	10/15/2000	1,690	701	2,440	8.6	.49	51.6	20.1	.05	24.5	50.4	126
Boggy Creek	01/10/2001	1,820	800	2,850	10.7	1.4	52.9	22.2	.06	21.6	44.8	172
Boggy Creek	[1]05/25/2001	1,170	801	3,010	11.4	.54	55.3	20.2	.04	22.3	35.4	136
Mean		1,760	840	2,904	10	.70	55	22	.05	27	41	148
C_v		.20	.13	.14	.10	.41	.11	.08	.17	.24	.17	.12
Williamson Creek	03/17/2000	1,830	2,850	3,680	15.5	1.4	87.6	42.2	na	52.3	78.5	466
Williamson Creek	05/01/2000	1,340	1,070	2,750	12.4	1.5	65.4	28.0	.07	32.4	57.2	231
Williamson Creek	11/03/2000	1,310	894	2,790	11.7	.80	57.3	24.8	.08	26.4	65.9	197
Williamson Creek	05/26/2001	1,040	1,110	3,650	13.1	.83	71.0	31.4	.07	29.0	65.3	292
Williamson Creek	[1]10/11/2001	949	901	3,180	11.3	.54	69.7	20.9	na	24.8	48.9	210
Williamson Creek	[1]03/19/2002	947	938	2,960	11.5	.78	58.2	24.8	na	24.2	55.8	237
Williamson Creek	02/20/2003	1,080	951	4,120	13.8	.55	64.0	24.0	.11	31.0	42.9	175
Williamson Creek	03/25/2003	2,200	1,290	2,780	10.9	.55	61.6	27.2	.06	25.1	48.9	241
Williamson Creek	11/17/2003	1,030	1,030	3,550	11.1	.70	65.2	30.5	.07	23.3	61.4	269
Mean		1,303	1,226	3,273	12	.84	67	28	.08	30	58	258
C_v		.34	.51	.15	.12	.42	.14	.22	.23	.30	.19	.33
TEC		--	--	--	9.79	.99	43.4	31.6	.18	22.7	35.8	121
PEC		--	--	--	33	4.98	111	149	1.06	48.6	128	459

[1] Value is average of duplicates except for SSC and sand content.

[2] Grab sample of small event, does not represent a storm composite; not included in computations of mean and C_v.

[3] Anomalous value; not reported or used in statistical analyses.

Appendix 3.2. Organochlorine compound concentrations associated with suspended sediment in composite rain event samples.

[In micrograms per kilogram; mean value listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
PCB, polychlorinated biphenyls; <, less than; E, estimated; C_v, coefficient of variation; --, not analyzed or not applicable; TEC, threshold effect concentration (MacDonald and others, 2000); PEC, probable effect concentration (MacDonald and others, 2000)]

Site	Date	Time	Dieldrin	Chlordane	DDD	DDE	DDT	Total DDT	PCB Aroclor 1242	PCB Aroclor 1254	PCB Aroclor 1260	Total PCBs
Barton 71	05/26/1999	2300	<12	<120	<12	<12	<12	<36	<120	<120	<120	<360
Barton 71	05/01/2000	2255	<6.3	<63	<6.3	<6.3	<117	<130	<63	<63	<63	<190
Barton 71	06/09/2000	0930	<.5	<5.5	<.5	<.5	<.5	<1.5	<5.5	<5.5	<5.5	<17
Barton 71	11/02–03/2000	1900	<2.5	<25	E1.7	E2.8	E7.6	E12	<25	<25	<25	<75
Barton 71	05/06/2001	2320	<2.0	<20	<2.0	E.7	E1.5	E2.2	<20	<20	<20	<60
Barton 71	11/15/2001	0945	<2.5	<25	<2.5	E.45	<2.5	E.45	<25	<25	<25	<75
Barton 71	06/30/2002	0505–1735	<2.0	<20	<2.0	E.28	E2.3	E2.6	<20	<20	<20	<60
Barton 71	02/20/2003	0920–0430	<1.5	<15	<1.5	<1.5	<1.5	<4.5	<15	<15	<15	<45
Barton 71	06/04/2003	0220	<3.5	<35	<3.5	E.99	<3.5	E.99	<35	<35	<35	<110
Barton 71	02/29/2004	1105	<100	<1,000	<100	<100	<100	<300	<1,000	<1,000	<1,000	<3,000
Mean			--	--	--	.80	--	2.5	--	--	--	--
C_v			--	--	--	.92	--	1.4	--	--	--	--
Barton Above	05/18/1999	0100	<1.9	E17	2.3	2.6	11	E16	<19	<19	<19	<57
Barton Above	05/26/1999	2100	<4.3	48	4.6	7.6	13	E25	<43	<43	<43	<130
Barton Above	05/02/2000	0135	<8.6	E39	<6.8	E7.1	<140	E7.1	<68	<68	<68	<200
Barton Above	06/09/2000	2055	<1.1	<11	<1.1	<1.1	<50	<52	<11	<11	<11	<33
Barton Above	11/03/2000	1940–0520	<18	<180	<18	E10	E12	E22	<180	<180	<180	<540
Barton Above	05/06/2001	2135	<10	<100	<10	11	10	E21	<100	<100	<100	<300
Barton Above	08/26/2001	2235	<7.0	<70	<7.0	E4.4	<7.0	E4.4	<70	<70	<70	<210
Barton Above	11/15/2001	1535	2.1	26	<1.5	6.1	9.8	E16	<15	<15	<15	<45
Barton Above	[1]06/30/2002	0455–2355	<7.8	<78	<7.8	E.92	<7.8	E.92	<78	<78	<78	<230
Barton Above	[1]02/20/2003	0510–0945	<7.0	<70	<7.0	<7.0	<7.0	<21	<70	<70	<70	<210
Barton Above	01/17/2004	0600	<130	<1,300	<130	<130	<130	<390	<1,300	<1,300	<1,300	<3,900
Mean			--	--	--	4.7	10	12	--	--	--	--
C_v			--	--	--	.73	.14	.60	--	--	--	--
Barton Springs	05/27/1999	0820	<.9	<8.9	<.9	<.9	<.9	<2.7	<8.9	<8.9	<8.9	<27
Barton Springs	05/01/2000	1800	<1.3	<13	<1.3	<1.3	<21	<24	<13	<13	<13	<39
Barton Springs	05/02/2000	1400	<1.6	<16	<1.6	<1.6	<96	<99	<16	<16	<16	<47
Barton Springs	06/09/2000	2050	<2.4	<24	<2.4	<2.4	<2.4	<7.3	<24	<24	<24	<73
Barton Springs	06/10/2000	1020	<1.9	<19	<1.9	<1.9	<1.9	<5.8	<19	<19	<19	<58
Barton Springs	11/03/2000	1030	<5.0	<50	<5.0	E1.0	<5.0	E1.0	<50	<50	<50	<150
Barton Springs	05/07/2001	0940	<6.0	<60	<6.0	<6.0	<6.0	<18	<60	<60	<60	<180
Barton Springs	08/27/2001	1000	<10	<100	<10	<10	<10	<30	<100	<100	<100	<300
Barton Springs	11/16/2001	1130	<6.0	<60	<6.0	E1.3	<6.0	E1.3	<60	<60	<60	<180
Barton Springs	06/30/2002	1650	<55	<550	<55	<55	<55	<170	<550	<550	<550	<1,700
Barton Springs	02/20/2003	2240	<20	<200	<20	<20	<20	<60	<200	<200	<200	<600
Barton Springs	01/17/2004	0815	<2.5	<25	<2.5	<2.5	<2.5	<7.5	<25	<25	<25	<75
Barton Springs	06/09/2004	0900	<10	--	<52	<32	<21	<110	<110	<110	<110	<330
Mean			--	--	--	--	--	--	--	--	--	--
C_v			--	--	--	--	--	--	--	--	--	--

Footnote at end of table.

Appendix 3.2. Organochlorine compound concentrations associated with suspended sediment in composite rain event samples—Continued.

Site	Date	Time	Dieldrin	Chlor-dane	DDD	DDE	DDT	Total DDT	PCB Aroclor 1242	PCB Aroclor 1254	PCB Aroclor 1260	Total PCBs
Shoal Creek	[2]03/12/1999	1200	<4.8	E36	8.0	10	7.6	26	<48	<48	<48	<145
Shoal Creek	03/18/1999	2120–0144	<1.1	E8.4	E2.6	6.5	4.2	13	<11	<11	<11	<33
Shoal Creek	04/26/1999	0910–1347	E3.3	E11	<3.9	E6.8	E5.4	E12	<39	E4.3	E2.5	E6.8
Shoal Creek	10/18/1999	1520–1620	<24	E250	<24	E230	E71	E300	<240	E170	<240	E170
Shoal Creek	10/30/1999	1030–1510	E1.2	<19	<1.9	E3.0	E2.4	E5.4	<19	<19	<19	<58
Shoal Creek	01/07/2000	1700	E2.0	E22	E1.7	E5.3	E16	E23	E19	E14	<13	E33
Shoal Creek	03/17/2000	0410–1150	E3.5	E56	E1.3	E9.3	E19	E30	<27	E7.1	E4.9	E12
Shoal Creek	10/15/2000	1945	E5.9	53	E7.9	22	24	E54	<35	E11	E10	E21
Shoal Creek	04/23/2001	1300	<88	<880	<88	E49	<88	E49	<880	<880	<880	<2,600
Mean			--	62	3.0	41	19	54	--	28	--	34
C_v			--	1.4	.83	1.9	1.2	1.7	--	1.9	--	1.5
Boggy Creek	03/18/1999	2200	E1.8	E11	E2.5	E20	E14	E36	<13	<13	<13	<39
Boggy Creek	05/10/1999	0500	E2.2	E22	<1.1	E22	E6.5	E28	<11	E6.5	E5.5	E12
Boggy Creek	10/30/1999	1130	<1.8	<18	<1.8	E9.6	E9.6	E19	<18	<18	<18	<54
Boggy Creek	01/07/2000	1700	<1.5	E18	E51	E16	E20	E87	<15	<15	<15	<45
Boggy Creek	02/23/2000	0100	<1.5	19	6.2	31	25	62	17	14	16	47
Boggy Creek	10/15–16/2000	2105–0110	E3.5	E47	E22	E70	E65	E160	<39	E15	E23	E37
Boggy Creek	01/10/2001	0940–1400	<7.4	E47	E2.1	E40	E30	E72	<74	E10	E9.0	E19
Boggy Creek	05/25/2001	0205	<9.0	<90	<9.0	E2.4	<9	E2.4	<90	<90	<90	<270
Mean			--	24	11	26	22	58	--	10	11	17
C_v			--	.60	1.7	.81	.89	.85	--	.28	.55	1.2
Williamson Creek	03/17/2000	0315	<6.1	<61	<6.1	<6.1	<6.5	<19	<61	<61	<61	<180
Williamson Creek	05/01/2000	0400	1.4	E9.0	<.7	2.0	<190	E2.0	17	E14	11	E42
Williamson Creek	11/03/2000	1410	<5.5	E55	<6.3	E3.7	<5.9	E3.7	<40	<40	<40	<120
Williamson Creek	05/26/2001	1245	<9.5	<95	<9.5	E1.9	<9.5	E1.9	<95	<95	<95	<280
Williamson Creek	10/11/2001	0600	E2.8	E34	<11	19	<11	E19	E43	<110	<110	E43
Williamson Creek	03/19/2002	2200	<8.0	<80	<8.0	E5.9	<17	E5.9	<80	<80	<80	<240
Williamson Creek	02/20/2003	0420	<7.0	E50	<7.0	E2.6	<7.0	E2.6	<70	<70	<70	<210
Williamson Creek	03/25/2003	1950	<13	E68	<13	<13	E3.3	E3.3	<130	<130	<130	<390
Williamson Creek	11/17/2003	1555	<14	E89	<14	<14	<14	<41	<140	<140	<140	<420
Mean			--	45	--	4.9	--	6.0	--	--	--	--
C_v			--	.53	--	1.1	--	1.1	--	--	--	--
TEC			1.9	3.24	4.88	3.16	4.16	5.28	--	--	--	59.8
PEC			61.8	17.6	28.0	31.3	62.9	572	--	--	--	676

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[1] Value is average of duplicates.

[2] Grab sample of small event, does not represent a storm composite; not included in computations of mean and C_v.

Appendix 3.3. Polycyclic aromatic hydrocarbon concentrations associated with suspended sediment in composite rain event samples.

[In micrograms per kilogram; mean value listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
PAH, polycyclic aromatic hydrocarbon; $\Sigma\text{PAH}_{\text{SQG}}$, total PAHs used for consensus-based sediment-quality guidelines (Ingersoll and others, 2000); E, estimated; <, less than; C_v, coefficient of variation; TEC, threshold effect concentration (MacDonald and others, 2000); PEC, probable effect concentration (MacDonald and others, 2000); 2+3/comb, ratio of 2- and 3-ringed PAHs to combustion PAHs]

Site	Date	Time	Total PAH	$\Sigma\text{PAH}_{\text{SQG}}$	Anthracene	Fluorene	Naphthalene	Phenanthrene
Barton 71	05/26/1999	2300	730	280	E17	<160	E14	E19
Barton 71	05/01/2000	2255	820	232	<170	E16	E3.2	E17
Barton 71	06/09/2000	0930	96	30	E.9	E.3	E2.0	E2.7
Barton 71	11/2–03/2000	1900	510	186	E7.0	<30	E4.4	E12
Barton 71	05/06/2001	2320	720	214	E7.3	E5.3	E2.3	E13
Barton 71	11/15/2001	0945	290	132	E7.2	E6.4	E4.4	E12
Barton 71	06/30/2002	0505–1735	290	74	E4.5	<25	E2.8	E9.8
Barton 71	02/20/2003	0920–0430	350	122	E4.5	E3.7	E6.3	E11
Barton 71	06/04/2003	0220	730	226	E7.5	<40	<40	E21
Barton 71	02/29/2004	1105	5,600	2,220	E190	<970	<970	E98
Mean			1,020	371	27	5.3	4.7	22
C_v			1.6	1.8	2.2	.90	.73	1.3
Barton Above	05/18/1999	0100	8,000	3,810	E38	E11	E6.4	330
Barton Above	05/26/1999	2100	21,000	10,200	E100	E24	E18	800
Barton Above	05/02/2000	0135	23,000	8,580	E75	E39	E26	790
Barton Above	06/09/2000	2055	5,300	2,540	E20	E5.8	E6.0	210
Barton Above	11/03/2000	1940–0520	16,000	6,320	E95	E33	E19	410
Barton Above	05/06/2001	2135	34,000	14,000	180	E57	E21	960
Barton Above	08/26/2001	2235	9,400	4,190	60	E23	E12	300
Barton Above	11/15/2001	1535	40,000	19,900	250	73	16	1,600
Barton Above	[1]06/30/2002	0455–2355	1,400	614	E15	E7.9	E6.4	E40
Barton Above	[1]02/20/2003	0510–0945	4,300	1,960	E28	E14	E16	130
Barton Above	01/17/2004	0600	50,000	22,400	E410	E230	<1,300	1,500
Mean			19,000	8,600	120	47	15	640
C_v			.83	.86	1.1	1.4	.45	.84
Barton Springs	05/27/1999	1440	86	43	E1.1	<11	E.4	E3.6
Barton Springs	05/01/2000	1800	390	172	<56	<56	E1.8	E12
Barton Springs	05/02/2000	1200	480	167	<29	E4.3	E1.1	E12
Barton Springs	06/09/2000	0200	200	71	E1.9	E.2	E1.2	E3.8
Barton Springs	06/10/2000	1020	230	95	E2.8	E.6	E4.5	E5.0
Barton Springs	11/03/2000	1030	390	136	E5.9	E1.6	E2.8	E7.3
Barton Springs	05/07/2001	0940	680	281	E22	E14	E2.7	E14
Barton Springs	08/27/2001	1314	980	503	E20	<130	E5.6	E30
Barton Springs	11/16/2001	1130	530	278	E17	E14	E3.1	E17
Barton Springs	06/30/2002	1650–2330	450	247	E21	<130	E6.6	E12
Barton Springs	02/20/2003	2120–0105	2,100	929	E44	E24	E23	E50
Barton Springs	01/17/2004	0815	630	237	E7.1	E4.1	<25	E11
Barton Springs	06/09/2004	0900	370	370	<220	<220	<220	<220
Mean			600	271	14	6.9	4.5	15
C_v			.85	.86	.94	1.1	1.3	.87

Footnote at end of table.

Appendix 3.3. Polycyclic aromatic hydrocarbon concentrations associated with suspended sediment in composite rain event samples—Continued.

Site	Date	Time	Total PAH	ΣPAH_{SQG}	Anthra-cene	Fluorene	Naphtha-lene	Phenan-threne
Shoal Creek	[2]03/12/1999	1200	24,300	11,400	157	E57	67	928
Shoal Creek	03/18/1999	2120–0144	21,000	8,040	E87	E35	E15	E920
Shoal Creek	04/26/1999	1347	15,000	6,610	E58	E21	E18	640
Shoal Creek	10/18/1999	1520–1620	57,000	27,700	350	E93	E120	2,100
Shoal Creek	10/30/1999	1030–1510	9,800	4,070	E40	E14	E13	380
Shoal Creek	01/07/2000	1700	15,000	6,650	51	23	E14	610
Shoal Creek	03/17/2000	0410–1150	19,000	8,780	E100	E29	E17	960
Shoal Creek	10/15/2000	1945	37,000	18,500	140	58	E22	1,400
Shoal Creek	04/23/2001	1300	110,000	43,900	800	E290	E130	3,200
Mean			36,000	15,000	200	70	43	1,300
C_v			.96	.87	1.3	1.3	1.1	.75
Boggy Creek	03/18/1999	2200	12,000	4,310	73	27	15	570
Boggy Creek	05/10/1999	0500	9,600	3,890	47	14	12	490
Boggy Creek	10/30/1999	1130	14,000	2,240	32	E11	E14	260
Boggy Creek	01/07/2000	1700	13,000	2,570	30	E13	E10	260
Boggy Creek	02/23/2000	0100	19,000	6,660	77	33	33.1	808
Boggy Creek	10/15/2000	2105	13,000	6,430	66	18	13	470
Boggy Creek	01/10/2001	0940–1015	30,000	10,800	150	41	27	840
Boggy Creek	05/25/2001	0205	4,100	1,620	68	22	9.0	150
Mean			14,000	4,800	68	22	17	480
C_v			.53	.63	.55	.48	.52	.53
Williamson Creek	03/17/2000	0315	120,000	57,900	340	61	11	4,300
Williamson Creek	05/01/2000	0400	79,000	34,600	230	67	21	3,000
Williamson Creek	11/02/2000	1410–1535	210,000	98,200	610	190	50	6,300
Williamson Creek	05/26/2001	1245	110,000	45,800	360	100	E44	2,900
Williamson Creek	10/11/2001	0505	97,000	44,000	400	110	E46	2,700
Williamson Creek	03/19/2002	2200	28,000	7,530	200	340	110	780
Williamson Creek	02/20/2003	0420–2015	110,000	56,700	400	100	E38	3,200
Williamson Creek	03/25/2003	1950	110,000	52,200	440	140	E77	3,500
Williamson Creek	11/17/2003	1555	210,000	94,900	720	220	E59	E5,400
Mean			120,000	55,000	410	150	50	3,600
C_v			.48	.52	.41	.61	.58	.45
TEC			--	1,610	57.2	77	176	204
PEC			--	22,800	845	536	561	1,170

Footnote at end of table.

Appendix 3.3. Polycyclic aromatic hydrocarbon concentrations associated with suspended sediment in composite rain event samples—Continued.

Site	Date	Time	Benz[*a*]-anthracene	Benzo[*a*]-pyrene	Chrysene	Fluor-anthene	Pyrene	2+3/comb
Barton 71	05/26/1999	2300	E26	E42	E23	E37	E40	1.0
Barton 71	05/01/2000	2255	E27	E37	E29	E32	E29	.7
Barton 71	06/09/2000	0930	E2.5	E2.9	E3.6	E5.5	E4.4	1.4
Barton 71	11/2–03/2000	1900	E14	E29	33	40	34	.5
Barton 71	05/06/2001	2320	26	27	29	46	36	.8
Barton 71	11/15/2001	0945	E13	E16	E9.2	E16	E14	1.3
Barton 71	06/30/2002	0505–1735	E6.6	E9.4	E11	E16	E14	1.6
Barton 71	02/20/2003	0920–0430	E9.2	E14	E14	21	E17	1.5
Barton 71	06/04/2003	0220	E20	E25	E31	46	43	1.2
Barton 71	02/29/2004	1105	E240	E350	E120	E290	E260	.9
Mean			38	55	30	58	49	1.1
C_v			1.9	1.9	1.1	1.6	1.5	.34
Barton Above	05/18/1999	0100	300	490	640	1,000	830	.10
Barton Above	05/26/1999	2100	810	1,300	1,700	2,700	2,300	.09
Barton Above	05/02/2000	0135	670	1,200	1,700	2,100	1,700	.09
Barton Above	06/09/2000	2055	240	350	460	640	510	.11
Barton Above	11/03/2000	1940–0520	440	990	1,200	1,600	1,200	.11
Barton Above	05/06/2001	2135	1,600	2,200	2,100	3,500	2,800	.13
Barton Above	08/26/2001	2235	350	280	760	1,100	800	.12
Barton Above	11/15/2001	1535	2,300	2,800	3,100	5,100	3,900	.11
Barton Above	[1]06/30/2002	0455–2355	E51	E78	E100	150	120	.25
Barton Above	[1]02/20/2003	0510–0945	190	280	320	500	390	.21
Barton Above	01/17/2004	0600	1,700	3,000	4,000	5,800	4,400	.15
Mean			780	1,200	1,500	2,200	1,700	.13
C_v			.94	.86	.84	.86	.85	.38
Barton Springs	05/27/1999	1440	E3.4	E5.1	E6.4	E10	E8.8	.20
Barton Springs	05/01/2000	1800	E12	E25	E31	39	E33	.12
Barton Springs	05/02/2000	1200	E13	E24	31	34	E29	.14
Barton Springs	06/09/2000	0200	E6.7	E9.8	E11	E13	E12	.53
Barton Springs	06/10/2000	1020	E7.8	E13	E14	E17	E15	.46
Barton Springs	11/03/2000	1030	E13	E25	E21	E24	E19	.50
Barton Springs	05/07/2001	0940	E36	E38	E26	E46	E41	.37
Barton Springs	08/27/2001	1314	E53	E86	E59	E98	E81	.16
Barton Springs	11/16/2001	1130	E34	E48	E20	E41	E36	.31
Barton Springs	06/30/2002	1650–2330	E24	E38	E25	E38	E35	.11
Barton Springs	02/20/2003	2120–0105	E68	E130	E120	E180	E150	.42
Barton Springs	01/17/2004	0815	E20	35	E39	E49	43	.31
Barton Springs	06/09/2004	0900	<220	E150	<220	E110	E100	0
Mean			24	48	33	54	47	.28
C_v			.82	.95	.92	.90	.88	.61

Footnote at end of table.

Appendix 3.3. Polycyclic aromatic hydrocarbon concentrations associated with suspended sediment in composite rain event samples—Continued.

Site	Date	Time	Benz[*a*]-anthracene	Benzo[*a*]-pyrene	Chrysene	Fluor-anthene	Pyrene	2+3/comb
Shoal Creek	[2]03/12/1999	1200	699	627	1,807	2,651	1,687	0.23
Shoal Creek	03/18/1999	2120–0144	E680	E900	E1,500	2,000	E1,600	.11
Shoal Creek	04/26/1999	1347	440	650	1,200	1,900	1,500	.13
Shoal Creek	10/18/1999	1520–1620	2,400	3,100	5,300	7,300	5,900	.11
Shoal Creek	10/30/1999	1030–1510	240	430	700	1,200	900	.15
Shoal Creek	01/07/2000	1700	540	820	1,100	1,900	1,400	.12
Shoal Creek	03/17/2000	0410–1150	620	980	1,600	2,400	1,800	.16
Shoal Creek	10/15/2000	1945	1,600	2,500	3,800	4,700	3,800	.10
Shoal Creek	04/23/2001	1300	3,200	6,200	7,600	12,000	8,700	.18
Mean			1,200	1,900	2,800	4,100	3,200	.14
C_v			.88	1.0	.87	.88	.86	.28
Boggy Creek	03/18/1999	2200	450	570	880	1,400	1,100	.15
Boggy Creek	05/10/1999	0500	340	490	940	1,300	1,100	.14
Boggy Creek	10/30/1999	1130	160	700	430	830	640	.22
Boggy Creek	01/07/2000	1700	240	540	500	870	660	.14
Boggy Creek	02/23/2000	0100	693	1,080	1,470	2,100	1,620	.22
Boggy Creek	10/15/2000	2105	610	860	1,300	1,700	1,300	.11
Boggy Creek	01/10/2001	0940–1015	990	1,400	1,800	3,000	2,200	.18
Boggy Creek	05/25/2001	0205	200	190	250	360	270	.31
Mean			460	650	940	1,430	1,100	.18
C_v			.63	.65	.57	.57	.55	.35
Williamson Creek	03/17/2000	0315	2,800	6,500	12,000	17,000	13,000	.08
Williamson Creek	05/01/2000	0400	1,800	4,000	6,200	10,000	8,000	.09
Williamson Creek	11/02/2000	1410–1535	7,600	14,000	18,000	27,000	21,000	.08
Williamson Creek	05/26/2001	1245	3,100	6,400	9,800	13,000	9,400	.08
Williamson Creek	10/11/2001	0505	3,400	5,800	8,700	12,000	8,800	.09
Williamson Creek	03/19/2002	2200	560	680	720	1,700	1,100	2.0
Williamson Creek	02/20/2003	0420–2015	5,600	7,000	10,000	16,000	13,000	0
Williamson Creek	03/25/2003	1950	3,600	6,300	9,300	E15,000	12,000	.10
Williamson Creek	11/17/2003	1555	9,000	13,000	E17,000	E26,000	E20,000	.07
Mean			4,200	7,100	10,000	15,000	12,000	.29
C_v			.66	.58	.51	.50	.53	2.2
TEC			108	150	166	423	195	--
PEC			1,050	1,450	1,290	2,230	1,520	--

[1] Value is average of duplicates.

[2] Grab sample of small event, does not represent a storm composite; not included in computations of mean and C_v.

Appendix 4—Loads Associated With Suspended Sediment

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Appendix 4.1. Metals loads associated with suspended sediment.

[In grams; mean value listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
na, not analyzed because of insufficient sediment mass; --, not applicable]

Site	Date	Time	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc
Barton 71	05/26/1999	2300	163	7.9	970	303	571	0.9	342	2,050
Barton 71	05/01/2000	2255	3.4	1.0	16.4	9.3	14.5	0	15.2	136
Barton 71	[1]06/09/2000	0930	1,540	65.7	7,000	2,410	2,950	5.6	4,010	10,800
Barton 71	11/02–03/2000	1900	392	15.8	1,910	655	659	1.2	940	2,750
Barton 71	05/06/2001	2320	389	12.4	1,810	652	,711	2.1	1,130	3,630
Barton 71	11/15/2001	0945	55,800	2,610	313,000	118,000	156,000	407	178,000	557,000
Barton 71	[1]06/30/2002	0505–1735	6,490	351	33,500	14,900	12,600	27.7	18,600	60,300
Barton 71	[1]02/20/2003	0920–0430	6,810	207	25,500	9,220	12,800	35.9	20,500	51,600
Barton 71	06/04/2003	0220	173	5.8	766	292	372	.6	533	1,360
Barton 71	02/29/2004	1105	30.9	1.3	147	62.4	67.7	0	37.5	351
Barton Above	05/18/1999	0100	129	5.1	709	225	590	.7	255	1,760
Barton Above	05/26/1999	2100	26.8	1.2	143	45.5	125	.2	52.7	477
Barton Above	05/02/2000	0135	8.0	1.8	34.6	15.6	38.5	.1	20.5	163
Barton Above	06/09/2000	2055	820	49.6	4,310	1,510	2,510	2.8	2,660	21,100
Barton Above	11/03/2000	1940–0520	162	7.7	674	247	568	.3	282	2,100
Barton Above	05/06/2001	2135	78.6	3.6	375	187	444	na	187	1,720
Barton Above	08/26/2001	2235	257	9.0	1,121	340	585	.7	452	2,840
Barton Above	[1]11/15/2001	1535	9,460	245	38,300	12,800	26,700	81.5	16,800	86,000
Barton Above	06/30/2002	0455–2355	2,150	87.4	11,100	4,940	4,620	10.9	5,790	29,200
Barton Above	2/20/2003	0510–0945	4,510	153	19,100	14,900	11,100	25.9	14,300	50,000
Shoal Creek	[2]03/12/1999	--	--	--	--	--	--	--	--	--
Shoal Creek	03/18/1999	2120–0144	4,450	246	25,500	8,980	14,900	18.9	10,200	66,700
Shoal Creek	03/28/1999	0045	386	6.5	2,240	562	797	1.3	1,060	3,300
Shoal Creek	04/26/1999	1347	228	10.8	1,470	584	956	1.3	529	3,690
Shoal Creek	10/18/1999	1520–1620	2,720	.7	15.6	16.7	40.9	na	7.1	115
Shoal Creek	10/30/1999	1030–1510	987	49.4	7,670	2,320	3,720	na	2,710	15,300
Shoal Creek	[1]01/07/2000	1700	2,220	101	15,500	5,170	8,170	11.3	8,210	31,800
Shoal Creek	03/17/2000	0410–1150	164	9.5	1,390	443	564	.6	499	3,100
Shoal Creek	10/15/2000	1945	4,340	321	26,500	9,000	16,600	27.1	10,700	53,500
Shoal Creek	04/23/2001	1300	9.1	.9	49.8	44.8	74.1	na	19.4	261
Boggy Creek	03/18/1999	2200	4,680	296	24,300	11,000	23,300	23.2	11,100	77,800
Boggy Creek	05/10/1999	0500	1,400	74.2	7,060	3,190	5,620	8.2	3,060	20,800
Boggy Creek	10/30/1999	1130	599	34.7	3,580	1,490	2,030	2.5	1,650	8,380
Boggy Creek	01/07/2000	1700	1,820	154	10,800	4,520	7,800	8.5	8,600	28,500
Boggy Creek	[1]02/23/2000	0100	1,780	114	11,400	4,040	5,890	9.9	5,080	26,400
Boggy Creek	10/15/2000	2105	4,830	276	29,100	11,300	28,400	28.8	13,800	71,200
Boggy Creek	01/10/2001	0940–1015	979	126	4,840	2,030	4,100	5.0	1,980	15,700
Boggy Creek	[1]05/25/2001	0205	256	12.2	1,250	455	,798	.9	503	3,070
Williamson Creek	03/17/2000	0315	4.2	.4	24.0	11.5	21.5	na	14.3	128
Williamson Creek	05/01/2000	0400	491	58.2	2,588	1,108	2,263	2.8	1,282	9,140
Williamson Creek	11/03/2000	1410–1535	225	15.3	1,100	476	1,266	1.5	507	37,83
Williamson Creek	05/26/2001	1245	30.1	1.9	163	72.1	150	.2	66.6	671
Williamson Creek	[1]10/11/2001	0505	46.5	2.2	287	85.7	201	na	102	862
Williamson Creek	03/19/2002	2200	52.8	3.6	268	114.3	257	na	111	1,092
Williamson Creek	02/20/2003	0420–2015	1,715.3	68.7	7,955	2,983	5,332	13.7	3,853	21,752
Williamson Creek	03/25/2003	1950	7.6	.4	42.8	18.9	34.0	0	17.5	168
Williamson Creek	11/17/2003	1555	199	12.6	1,170	547	1,102	1.3	418	482

[1] Value is average of duplicates.

[2] Grab sample; load not computed.

Appendix 4.2. Organochlorine compound loads associated with suspended sediment.

[In milligrams; mean value listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
PCB, polychlorinated biphenyls; NC, not computed because compound not detected; --, not applicable]

Site	Date	Time	Dieldrin	Chlordane	DDD	DDE	DDT	Total DDT	PCB Aroclor 1242	PCB Aroclor 1254	PCB Aroclor 1260	Total PCBs
Barton 71	05/26/1999	2300	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton 71	05/01/2000	2255	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton 71	06/09/2000	0930	NC	NC	NC	NC	0	NC	NC	NC	NC	NC
Barton 71	11/02–03/2000	1900	NC	NC	69	110	310	490	NC	NC	NC	NC
Barton 71	05/06/2001	2320	NC	NC	NC	37	80	120	NC	NC	NC	NC
Barton 71	11/15/2001	0945	NC	NC	NC	4,600	0	4,600	NC	NC	NC	NC
Barton 71	06/30/2002	0505–1735	NC	NC	NC	260	2,100	2,400	NC	NC	NC	NC
Barton 71	02/20/2003	0920–0430	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton 71	06/04/2003	0220	NC	NC	NC	21	NC	21	NC	NC	NC	NC
Barton 71	02/29/2004	1105	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton Above	05/18/1999	0100	NC	180	24	27	120	170	NC	NC	NC	NC
Barton Above	05/26/1999	2100	NC	38	5	6	29	35	NC	NC	NC	NC
Barton Above	05/02/2000	0135	NC	20	NC	4	NC	4	NC	NC	NC	NC
Barton Above	06/09/2000	2055	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton Above	11/03/2000	1940–0520	NC	NC	NC	100	120	220	NC	NC	NC	NC
Barton Above	05/06/2001	2135	NC	NC	NC	93	84	180	NC	NC	NC	NC
Barton Above	08/26/2001	2235	NC	NC	NC	78	0	78	NC	NC	NC	NC
Barton Above	11/15/2001	1535	1,700	21,000	NC	5,000	8,000	13,000	NC	NC	NC	NC
Barton Above	[1]06/30/2002	0455–2355	NC	NC	NC	NC	0	250	NC	NC	NC	NC
Barton Above	[1]02/20/2003	0510–0945	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton Above	01/17/2004	600	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Shoal Creek	[2]03/12/1999	--	--	--	--	--	--	--	--	--	--	--
Shoal Creek	03/18/1999	2120–0144	NC	3,100	560	2,100	1,200	4,800	NC	NC	NC	NC
Shoal Creek	04/26/1999	1347	65	220	NC	140	190	260	NC	87	51	138
Shoal Creek	10/18/1999	1520–1620	NC	56	NC	53	16	69	NC	39	NC	39
Shoal Creek	10/30/1999	1030–1510	110	NC	NC	280	230	500	NC	NC	NC	NC
Shoal Creek	01/07/2000	1700	450	5,000	380	1,200	23	5,100	4,300	3,300	NC	7,600
Shoal Creek	03/17/2000	0410–1150	51	820	19	140	280	440	NC	100	73	180
Shoal Creek	10/15/2000	1945	2,300	21,000	3,100	8,500	9,300	21,000	NC	4,300	3,900	8,100
Shoal Creek	04/23/2001	1300	NC	NC	NC	40	NC	40	NC	NC	NC	NC
Boggy Creek	03/18/1999	2200	840	5,200	1,200	9,100	6,400	17,000	NC	NC	NC	NC
Boggy Creek	05/10/1999	0500	300	3,000	NC	3,000	900	4,000	NC	890	760	1,600
Boggy Creek	10/30/1999	1130	NC	NC	NC	590	590	1,200	NC	NC	NC	NC
Boggy Creek	01/07/2000	1700	NC	3,800	11,000	3,400	4,300	18,000	NC	NC	NC	NC
Boggy Creek	02/23/2000	0100	NC	3,100	1,000	5,100	4,100	10,000	2,800	2,300	2,600	7,800
Boggy Creek	10/15/2000	2105	2,000	27,000	12,000	40,000	37,000	89,000	NC	8,500	13,000	21,000
Boggy Creek	01/10/2001	0940–1015	NC	4,300	190	3,700	2,700	6,600	NC	920	820	1,700
Boggy Creek	05/25/2001	0205	NC	NC	NC	54	NC	54	NC	NC	NC	NC
Williamson Creek	03/17/2000	0315	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Williamson Creek	05/01/2000	0400	55	360	NC	78	NC	78	78	670	550	450
Williamson Creek	11/02/2000	1410–1535	NC	1,100	NC	71	NC	71	71	NC	NC	NC
Williamson Creek	05/26/2001	1245	NC	NC	NC	4	NC	4	4	NC	NC	NC
Williamson Creek	10/11/2001	0505	12	140	NC	78	NC	78	78	180	NC	NC
Williamson Creek	03/19/2002	2200	NC	NC	NC	27	NC	27	27	NC	NC	NC
Williamson Creek	02/20/2003	0420–2015	NC	6,200	NC	320	NC	320	NC	NC	NC	NC
Williamson Creek	03/25/2003	1950	NC	47	NC	NC	2	2	2	NC	NC	NC
Williamson Creek	11/17/2003	1555	NC	1,600	NC	NC	NC	NC	NC	NC	NC	NC

[1] Value is average of duplicates.

[2] Grab sample; load not computed.

Appendix 4.3. Polycyclic aromatic hydrocarbon loads associated with suspended sediment.

[In milligrams; mean values listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
PAH, polycyclic aromatic hydrocarbon; ΣPAH_{SQG}, total PAHs used for consensus-based sediment-quality guidelines (Ingersoll and others, 2000); NC, not computed because compound not detected; --, not applicable]

Site	Date	Time	Total PAH	ΣPAH_{SQG}	Anthracene	Fluorene	Naphthalene	Phenanthrene	Benz[*a*]anthracene	Benzo[*a*]pyrene	Chrysene	Fluoranthene	Pyrene
Barton 71	05/26/1999	2300	11,000	4,100	250	NC	203	270	380	610	340	540	590
Barton 71	05/01/2000	2255	410	120	NC	8.0	1.6	8.4	13	18	15	NC	15
Barton 71	06/09/2000	0930	18,000	5,700	170	47	370	510	470	550	670	1,000	830
Barton 71	11/02–03/2000	1900	21,000	7,600	290	NC	180	500	580	1,200	1,300	1,600	1,400
Barton 71	05/06/2001	2320	39,000	11,000	390	280	120	720	1,400	1,500	1,600	2,500	1,900
Barton 71	11/15/2001	0945	3,000,000	1,300,000	73,000	65,000	45,000	120,000	130,000	160,000	94,000	170,000	140,000
Barton 71	06/30/2002	0505–1735	270,000	68,000	4,200	NC	2,600	9,000	6,100	8,700	10,000	15,000	13,000
Barton 71	02/20/2003	0920–0430	310,000	110,000	4,000	3,300	5,700	9,700	8,300	13,000	13,000	19,000	15,000
Barton 71	06/04/2003	0220	16,000	4,900	160	NC	NC	460	420	540	670	1,000	930
Barton 71	02/29/2004	1105	20,000	7,800	670	NC	NC	340	840	1,200	420	1,000	900
Barton Above	05/18/1999	0100	84,000	40,000	400	110	68	3,500	3,200	5,200	6,800	11,000	8,800
Barton Above	05/26/1999	2100	48,000	23,000	230	54	40	1,800	1,800	2,900	3,800	6,100	5,200
Barton Above	05/02/2000	0135	11,000	4,300	38	20	13	390	340	590	840	1,100	840
Barton Above	06/09/2000	2055	370,000	180,000	1,400	410	420	15,000	17,000	25,000	33,000	44,000	36,000
Barton Above	11/03/2000	1940–0520	160,000	63,000	950	330	190	4,100	4,400	10,000	12,000	16,000	12,000
Barton Above	05/06/2001	2135	290,000	120,000	1,500	480	180	8,100	13,000	18,000	18,000	30,000	24,000
Barton Above	08/26/2001	2235	170,000	74,000	1,100	410	210	5,200	6,200	10,000	13,000	19,000	14,000
Barton Above	11/15/2001	1535	32,000,000	16,000,000	200,000	60,000	13,000	1,300,000	1,900,000	2,300,000	2,500,000	4,100,000	3,200,000
Barton Above	[1]06/30/2002	0455–2355	390,000	170,000	4,000	2,200	1,700	11,000	14,000	21,000	28,000	41,000	33,000
Barton Above	[1]02/20/2003	0510–0945	2,800,000	1,300,000	18,000	9,000	10,000	86,000	120,000	180,000	200,000	320,000	250,000
Barton Above	01/17/2004	0600	82,000	36,000	660	370	NC	2,500	2,700	4,900	6,500	9,300	7,200
Shoal Creek	[2]03/12/1999	--	--	--	--	--	--	--	--	--	--	--	--
Shoal Creek	03/18/1999	2120–0144	7,700,000	2,900,000	32,000	13,000	5,400	330,000	250,000	330,000	540,000	750,000	580,000
Shoal Creek	04/26/1999	1347	310,000	130,000	1,200	420	360	13,000	8,900	13,000	25,000	38,000	30,000
Shoal Creek	10/18/1999	1520–1620	13,000	6,400	80	21	26	490	540	720	1,200	1,700	1,300
Shoal Creek	10/30/1999	1030–1510	920,000	380,000	3,700	1,300	1,200	36,000	23,000	40,000	65,000	120,000	85,000
Shoal Creek	01/07/2000	1700	3,400,000	150,000	12,000	5,300	3,300	140,000	120,000	190,000	250,000	430,000	330,000
Shoal Creek	03/17/2000	0410–1150	280,000	130,000	1,500	430	250	14,000	9,100	14,000	24,000	35,000	27,000
Shoal Creek	10/15/2000	1945	14,000,000	7,200,000	55,000	22,000	8,700	530,000	630,000	980,000	1,500,000	1,800,000	1,500,000
Shoal Creek	04/23/2001	1300	92,000	36,000	660	230	100	2,600	2,600	5,000	6,200	9,500	7,100
Boggy Creek	03/18/1999	2200	5,500,000	2,000,000	34,000	12,000	7,100	270,000	210,000	270,000	410,000	640,000	510,000
Boggy Creek	05/10/1999	0500	1,300,000	530,000	6,400	1,900	1,600	67,000	46,000	67,000	130,000	180,000	150,000
Boggy Creek	10/30/1999	1130	890,000	140,000	2,000	670	890	16,000	9,500	43,000	26,000	51,000	39,000
Boggy Creek	01/07/2000	1700	2,700,000	550,000	6,300	2,700	2,100	54,000	52,000	120,000	110,000	180,000	140,000
Boggy Creek	02/23/2000	0100	1,098,683	1,098,682	13,000	5,400	5,500	130,000	110,000	180,000	240,000	350,000	270,000
Boggy Creek	10/15/2000	2105	7,400,000	3,600,000	37,000	10,000	7,400	270,000	340,000	490,000	740,000	930,000	730,000
Boggy Creek	01/10/2001	0940–1015	2,700,000	990,000	17,000	3,800	2,500	77,000	91,000	130,000	160,000	270,000	200,000
Boggy Creek	05/25/2001	0205	92,000	37,000	1,500	510	200	3,400	4,400	4,300	5,600	8,000	6,100
Williamson Creek	03/17/2000	0315	33,000	16,000	93	17	3	1,200	770	1,800	3,300	4,800	3,600
Williamson Creek	05/01/2000	0400	3,100,000	1,300,000	9,200	2,700	830	120,000	72,000	160,000	250,000	410,000	310,000
Williamson Creek	11/02/2000	1410–1535	3,900,000	1,900,000	12,000	3,600	970	120,000	150,000	270,000	340,000	520,000	410,000
Williamson Creek	05/26/2001	1245	250,000	110,000	830	240	100	6,600	7,000	15,000	22,000	29,000	22,000
Williamson Creek	10/11/2001	0505	400,000	180,000	1,600	440	190	11,000	14,000	24,000	36,000	50,000	36,000
Williamson Creek	03/19/2002	2200	130,000	35,000	900	1,600	500	3,600	2,600	3,100	3,300	7,900	5,200
Williamson Creek	02/20/2003	0420–2015	14,000,000	7,100,000	49,000	12,000	4,700	400,000	700,000	870,000	1,200,000	2,000,000	1,600,000
Williamson Creek	03/25/2003	1950	78,000	36,000	310	94	53	2,500	2,500	4,400	6,500	11,000	8,000
Williamson Creek	11/17/2003	1555	3,800,000	1,700,000	13,000	3,900	1,100	97,000	160,000	230,000	300,000	470,000	360,000

[1] Value is average of duplicates.

[2] Grab sample; load not computed.

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Appendix 5—Yields Associated With Suspended Sediment

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Appendix 5.1. Metals yields associated with suspended sediment.

[In milligrams per acre; mean value listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
na, not analyzed because of insufficient sediment mass; --, not applicable]

Site	Date	Time	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc
Barton 71	05/26/1999	2300	2.8	0.1	16.9	5.3	10.0	0	6.0	35.8
Barton 71	05/01/2000	2255	.1	0	.3	.2	.3	0	.3	2.4
Barton 71	[1]06/09/2000	0930	26.7	1.1	122	42.0	51.5	.1	69.8	189
Barton 71	11/02–03/2000	1900	6.8	.3	33.3	11.4	11.5	0	16.4	47.9
Barton 71	05/06/2001	2320	6.8	.2	31.6	11.4	12.4	0	19.7	63.2
Barton 71	11/15/2001	0945	972	45.4	5,450	2,060	2,720	7.1	3,110	9,710
Barton 71	[1]06/30/2002	0505–1735	113	6.1	584	259	220	.5	323	1,050
Barton 71	[1]2/20/2003	0920–0430	119	3.6	444	161	223	.6	357	899
Barton 71	06/04/2003	0220	3.0	.1	13.4	5.1	6.5	0	9.3	23.7
Barton 71	02/29/2004	1105	.5	0	2.6	1.1	1.2	0	.7	6.1
Barton Above	05/18/1999	0100	1.7	.1	9.2	2.9	7.7	0	3.3	22.9
Barton Above	05/26/1999	2100	.3	0	1.9	.6	1.6	0	.7	6.2
Barton Above	05/02/2000	0135	.1	0	.4	.2	.5	0	.3	2.1
Barton Above	06/09/2000	2055	10.7	.6	56.0	19.7	32.6	0	34.5	274
Barton Above	11/03/2000	1940–0520	2.1	.1	8.8	3.2	7.4	0	3.7	27.3
Barton Above	05/06/2001	2135	1.0	0	4.9	2.4	5.8	na	2.4	22.4
Barton Above	08/26/2001	2235	3.3	.1	14.6	4.4	7.6	0	5.9	36.9
Barton Above	[1]11/15/2001	1535	123	3.2	498	166	347	1.1	218	1,120
Barton Above	06/30/2002	0455–2355	27.9	1.1	145	64.3	60.0	.1	75.3	380
Barton Above	2/20/2003	0510–0945	58.6	2.0	248	193	145	.3	187	650
Shoal Creek	[2]03/12/1999	--	--	--	--	--	--	--	--	--
Shoal Creek	03/18/1999	2120–0144	538	29.8	3,080	1,090	1,810	2.3	1,230	8,060
Shoal Creek	03/28/1999	0045	46.7	.8	271	67.9	96.4	.2	129	398
Shoal Creek	04/26/1999	1347	27.6	1.3	177	70.6	116	.2	64.0	446
Shoal Creek	10/18/1999	1520–1620	329	.1	1.9	2.0	4.9	na	.9	14.0
Shoal Creek	10/30/1999	1030–1510	119	6.0	928	280	450	na	327	1,850
Shoal Creek	[1]01/07/2000	1700	269	12.3	1,870	625	988	1.4	993	3,840
Shoal Creek	03/17/2000	0410–1150	19.9	1.2	168	53.5	68.2	.1	60.3	375
Shoal Creek	10/15/2000	1945	525	38.8	3,200	1,090	2,010	3.3	1,290	6,470
Shoal Creek	04/23/2001	1300	1.1	.1	6.0	5.4	9.0	na	2.3	31.6
Boggy Creek	03/18/1999	2200	551	34.9	2,870	1,300	2,770	2.7	1,300	9,170
Boggy Creek	05/10/1999	0500	165	8.7	832	376	662	1.0	361	2,450
Boggy Creek	10/30/1999	1130	70.5	4.1	421	175	240	.3	194	988
Boggy Creek	01/07/2000	1700	214	18.1	1,270	533	918	1.0	1,010	3,350
Boggy Creek	[1]02/23/2000	0100	210	13.4	1,350	476	694	1.2	598	3,110
Boggy Creek	10/15/2000	2105	570	32.6	3,430	1,340	3,350	3.4	1,630	8,390
Boggy Creek	01/10/2001	0940–1015	115	14.9	570	239	483	.6	233	1,850
Boggy Creek	[1]05/25/2001	0205	30.2	1.4	147	53.6	94.0	.1	59.2	361
Williamson Creek	03/17/2000	0315	.2	0	1.2	.6	1.1	na	.7	6.6
Williamson Creek	05/01/2000	0400	25.5	3.0	134	57.5	117	.1	66.5	474
Williamson Creek	11/03/2000	1410–1535	11.7	.8	57.1	24.7	65.7	.1	26.3	196
Williamson Creek	05/26/2001	1245	1.6	.1	8.5	3.7	7.8	0	3.5	34.8
Williamson Creek	[1]10/11/2001	0505	2.4	.1	14.9	4.5	10.4	na	5.3	44.7
Williamson Creek	[1]03/19/2002	2200	2.7	.2	13.9	5.9	13.3	na	5.8	56.7
Williamson Creek	2/20/2003	0420–2015	89.0	3.6	413	155	277	.7	200	1,130
Williamson Creek	03/25/2003	1950	.4	0	2.2	1.0	1.8	0	.9	8.7
Williamson Creek	11/17/2003	1555	10.3	.7	60.7	28.4	57.2	.1	21.7	251

[1] Value is average of duplicates.

[2] Grab sample; yield not computed.

Appendix 5.2. Organochlorine compound yields associated with suspended sediment.

[In micrograms per acre; mean value listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
PCB, polychlorinated biphenyls; NC, not computed because compound not detected; --, not applicable]

Site	Date	Time	Diel-drin	Chlor-dane	DDD	DDE	DDT	Total DDT	PCB Aroclor 1242	PCB Aroclor 1254	PCB Aroclor 1260	Total PCB
Barton 71	05/26/1999	2300	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton 71	05/01/2000	2255	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton 71	06/09/2000	0930	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton 71	11/02–03/2000	1900	NC	NC	1.2	2.0	5.4	8.6	NC	NC	NC	NC
Barton 71	05/06/2001	2320	NC	NC	NC	.7	1.4	2.0	NC	NC	NC	NC
Barton 71	11/15/2001	0945	NC	NC	NC	80	NC	80	NC	NC	NC	NC
Barton 71	06/30/2002	0505–1735	NC	NC	NC	4.5	37	42	NC	NC	NC	NC
Barton 71	02/20/2003	0920–0430	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton 71	06/04/2003	0220	NC	NC	NC	.4	NC	.4	NC	NC	NC	NC
Barton 71	02/29/2004	1105	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton Above	05/18/1999	0100	NC	2.3	.3	.4	1.5	2.2	NC	NC	NC	NC
Barton Above	05/26/1999	2100	NC	.5	.1	.1	.4	.5	NC	NC	NC	NC
Barton Above	05/02/2000	0135	NC	.3	NC	0	NC	0	NC	NC	NC	NC
Barton Above	06/09/2000	2055	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton Above	11/03/2000	none	NC	NC	NC	1.3	1.6	2.9	NC	NC	NC	NC
Barton Above	05/06/2001	2135	NC	NC	NC	1.2	1.1	2.3	NC	NC	NC	NC
Barton Above	08/26/2001	2235	NC	NC	NC	1.0	NC	1.0	NC	NC	NC	NC
Barton Above	11/15/2001	1535	22	276	NC	65	104	170	NC	NC	NC	NC
Barton Above	[1]06/30/2002	0455–2355	NC	NC	NC	3.3	NC	3.3	NC	NC	NC	NC
Barton Above	[1]02/20/2003	0510–0945	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Barton Above	01/17/2004	600	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Shoal Creek	[2]03/12/1999	1200	--	--	--	--	--	--	--	--	--	--
Shoal Creek	03/18/1999	2120–0144	NC	370	68	260	150	580	NC	NC	NC	NC
Shoal Creek	04/26/1999	1347	7.9	26	NC	17	23	31	NC	11	6.1	17
Shoal Creek	10/18/1999	1520–1620	NC	6.8	NC	6.4	2.0	8.3	NC	4.7	NC	4.7
Shoal Creek	10/30/1999	1030–1510	13	NC	NC	34	27	61	NC	NC	NC	NC
Shoal Creek	01/07/2000	1700	55	610	46	150	3	620	520	400	NC	910
Shoal Creek	03/17/2000	0410–1150	6.2	100	2	17	34	53	NC	13	8.8	21
Shoal Creek	10/15/2000	1945	280	2,500	370	1,000	1,100	2,500	NC	520	470	990
Shoal Creek	04/23/2001	1300	NC	NC	NC	4.8	NC	4.8	NC	NC	NC	NC
Boggy Creek	03/18/1999	2200	99	610	140	1,100	760	2,000	NC	NC	NC	NC
Boggy Creek	05/10/1999	0500	35	350	NC	350	110	460	NC	110	89	190
Boggy Creek	10/30/1999	1130	NC	NC	NC	70	70	140	NC	NC	NC	NC
Boggy Creek	01/07/2000	1700	NC	450	1,300	400	500	2,200	NC	NC	NC	NC
Boggy Creek	02/23/2000	0100	NC	370	120	600	490	1,200	330	270	310	910
Boggy Creek	10/15/2000	2105	230	3,100	1,500	4,700	4,300	10,000	NC	1,000	1,500	2,500
Boggy Creek	01/10/2001	0940–1015	NC	510	23	430	320	780	NC	110	97	210
Boggy Creek	05/25/2001	0205	NC	NC	NC	6.4	NC	6.4	NC	NC	NC	NC
Williamson Creek	03/17/2000	0315	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Williamson Creek	05/01/2000	0400	2.9	18	NC	4.0	NC	4.0	NC	NC	NC	NC
Williamson Creek	11/02/2000	1410–1535	NC	55	NC	3.7	NC	3.7	NC	NC	NC	NC
Williamson Creek	05/26/2001	1245	NC	NC	NC	.2	NC	.2	NC	NC	NC	NC
Williamson Creek	10/11/2001	0505	.6	7.3	NC	4.1	NC	4.1	NC	NC	NC	NC
Williamson Creek	03/19/2002	2200	NC	NC	NC	1.4	NC	1.4	NC	NC	NC	NC
Williamson Creek	02/20/2003	0420–2015	NC	323	NC	17	NC	17	NC	NC	NC	NC
Williamson Creek	03/25/2003	1950	NC	2.5	NC	NC	.1	NC	NC	NC	NC	NC
Williamson Creek	11/17/2003	1555	NC	83	NC	NC	NC	NC	NC	NC	NC	NC

[1] Value is average of duplicates.

[2] Grab sample; yield not computed.

Appendix 5.3. Polycyclic aromatic hydrocarbon yields associated with suspended sediment.

[In micrograms per acre; mean value listed for duplicate samples; most samples are flow-weighted composite samples; selected samples from Shoal and Boggy Creeks list calculated composite values based on discrete sample analyses.
PAH, polycyclic aromatic hydrocarbon; ΣPAH_{SQG}, total PAHs used for consensus-based sediment-quality guidelines (Ingersoll and others, 2000); NC, not computed because compound not detected; --, not applicable]

Site	Date	Time	Total PAH	ΣPAH_{SQG}	Anthra-cene	Fluor-ene	Naph-tha-lene	Phenan-threne	Benz-[*a*]-anthra-cene	Benzo-[*a*]-pyrene	Chry-sene	Fluor-an-thene	Pyrene
Barton 71	05/26/1999	2300	180	71	4.3	NC	3.5	4.7	6.7	11	5.9	9.4	10
Barton 71	05/01/2000	2255	7.2	2	NC	.1	0	.1	.2	.3	.3	NC	.3
Barton 71	06/09/2000	0930	310	99	2.9	.8	6.4	8.9	8.3	9.6	12	18	14
Barton 71	11/02–03/2000	1900	360	130	5.0	NC	3.1	8.7	10	21	23	29	24
Barton 71	05/06/2001	2320	670	200	6.8	4.9	2.1	12	24	25	27	43	34
Barton 71	11/15/2001	0945	52,000	23,000	1,300	1,100	780	2,100	2,300	2,800	1,600	2,900	2,500
Barton 71	06/30/2002	0505–1735	4,700	1,200	72	NC	45	160	110	150	180	250	230
Barton 71	02/20/2003	0920–0430	5,400	1,900	70	58	99	170	140	220	220	330	270
Barton 71	06/04/2003	0220	280	85	2.8	NC	NC	8.0	7.4	9.5	12	17	16
Barton 71	02/29/2004	1105	340	140	12	NC	NC	6.0	15	21	7.4	18	16
Barton Above	05/18/1999	0100	1,100	520	5.2	1.4	.9	46	41	67	88	140	110
Barton Above	05/26/1999	2100	620	290	3.0	.7	.5	23	23	37	50	80	67
Barton Above	05/02/2000	0135	150	56	.5	.3	.2	5.1	4.4	7.7	11	14	11
Barton Above	06/09/2000	2055	4,900	2,300	18.1	5.3	5.4	190	220	320	420	580	470
Barton Above	11/03/2000	none	2,100	820	12.4	4.2	2.5	53	57	130	150	203	160
Barton Above	05/06/2001	2135	3,700	1,500	19.9	6.2	2.3	110	170	240	230	390	310
Barton Above	08/26/2001	2235	2,200	970	13.8	5.3	2.7	68	81	130	180	250	190
Barton Above	11/15/2001	1535	420,000	210,000	2,700	780	170	17,000	24,000	30,000	32,000	54,000	41,000
Barton Above	[1]06/30/2002	0455–2355	5,100	2,200	52	28	23	140	180	280	370	530	420
Barton Above	[1]02/20/2003	0510–0945	39,000	16,000	240	120	131	1,100	1,600	2,400	2,700	4,200	3,300
Barton Above	01/17/2004	0600	1,100	473	8.6	4.8	NC	32	35	63	84	120	93
Shoal Creek	[2]03/12/1999	--	--	--	--	--	--	--	--	--	--	--	--
Shoal Creek	03/18/1999	2120–0144	940,000	360,000	3,800	1,500	650	40,000	30,000	39,000	65,000	90,000	70,000
Shoal Creek	04/26/1999	1347	37,000	16,000	140	50	44	1,600	1,100	1,600	3,100	4,600	3,600
Shoal Creek	10/18/1999	1520–1620	1,600	770	9.7	2.6	3.2	59	65	87	150	200	160
Shoal Creek	10/30/1999	1030–1510	110,000	46,000	450	160	150	4,300	2,700	4,900	7,900	14,000	10,000
Shoal Creek	01/07/2000	1700	420,000	180,000	1,400	640	400	17,000	15,000	23,000	30,000	52,000	40,000
Shoal Creek	03/17/2000	0410–1150	34,000	16,000	180	52	30	1,700	1,100	1,700	2,900	4,300	3,200
Shoal Creek	10/15/2000	none	1,700,000	870,000	6,700	2,700	1,100	64,000	76,000	120,000	180,000	220,000	180,000
Shoal Creek	04/23/2001	1300	11,000	4,300	79	28	13	320	310	610	750	1,100	860
Boggy Creek	03/18/1999	2200	640,000	240,000	4,000	1,400	840	31,000	24,000	31,000	48,000	75,000	60,000
Boggy Creek	05/10/1999	0500	160,000	63,000	760	230	190	7,900	5,500	7,900	15,000	21,000	18,000
Boggy Creek	10/30/1999	1130	100,000	16,000	230	79	110	1,900	1,100	5,100	3,100	6,000	4,600
Boggy Creek	01/07/2000	1700	320,000	64,000	740	320	240	6,400	6,100	14,000	12,000	22,000	16,000
Boggy Creek	02/23/2000	0100	370,000	129,000	1,500	640	640	16,000	13,000	21,000	29,000	41,000	31,000
Boggy Creek	10/15/2000	2105	870,000	430,000	4,400	1,200	870	31,000	40,000	57,000	87,000	110,000	87,000
Boggy Creek	01/10/2001	0940–1015	320,000	120,000	1,600	440	300	9,000	11,000	15,000	19,000	32,000	24,000
Boggy Creek	05/25/2001	0205	11,000	4,300	180	59	24	400	520	500	660	950	720
Williamson Creek	03/17/2000	0315	1,700	820	4.8	.9	.2	62	40	93	170	250	190
Williamson Creek	05/01/2000	0400	160,000	72,000	480	140	43	6,100	3,700	8,200	13,000	21,000	16,000
Williamson Creek	11/02/2000	1410–1535	200,000	98,000	610	190	50	6,300	7,600	14,000	18,000	27,000	21,000
Williamson Creek	05/26/2001	1245	13,000	5,500	43	12	5.2	341	360	760	1,200	1,500	1,100
Williamson Creek	10/11/2001	0505	21,000	9,400	85	23	9.8	590	720	1,200	1,900	2,600	1,900
Williamson Creek	03/19/2002	2200	6,600	1,800	47	81	26	190	130	160	170	410	270
Williamson Creek	02/20/2003	0420–2015	710,000	370,000	2,600	630	240	21,000	36,000	45,000	65,000	110,000	81,000
Williamson Creek	03/25/2003	1950	4,100	1,900	16	4.9	2.8	130	130	230	340	550	420
Williamson Creek	11/17/2003	1555	200,000	88,000	670	200	55	5,000	8,300	12,000	16,000	24,000	19,000

[1] Value is average of duplicates.

[2] Grab sample; yield not computed.

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Appendix 6—Concentrations Associated With Suspended Sediment in Discrete Rain Event Samples at Shoal and Boggy Creeks

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Appendix 6.1. Metals concentrations associated with suspended sediment in discrete rain event samples at Shoal and Boggy Creeks.

[In milligrams per kilogram, except as noted. SSC, suspended-sediment concentration; mg/L, milligrams per liter; --, not analyzed; na, not analyzed because of insufficient sediment mass]

Rain event	Date	Time	SSC (mg/L)	Sand content (percent)	Arsenic	Cadmium	Chromium	Copper	Mercury	Nickel	Lead	Zinc
						Shoal Creek						
SC1	03/12/1999	1200	83	--	13.7	3.16	69.6	69.2	0.20	30.8	196	607
SC1	03/18/1999	2120	3,181	--	11.7	.620	63.8	20.8	.04	25.1	41.7	152
SC1	03/18/1999	2147	1,774	--	11.8	.606	63.2	25.5	.05	27.4	43.6	181
SC1	03/18/1999	2230	819	--	12.5	.623	71.2	26.7	.06	29.1	40.8	189
SC1	03/19/1999	0144	116	--	12.5	.853	78.2	23.7	.05	28.7	38.6	198
SC2	04/26/1999	0910	59	0.4	10.8	3.02	82.0	170	.37	32.3	285	830
SC2	04/26/1999	1145	504	3.0	11.9	.606	53.5	33.0	.09	23.4	69.0	215
SC2	04/26/1999	1205	1,813	.3	12.2	.210	70.9	19.4	.03	25.7	27.9	102
SC2	04/26/1999	1247	390	.4	11.6	.385	75.4	23.8	.05	26.6	37.3	150
SC2	04/26/1999	1347	96	1.4	10.8	.595	74.3	28.2	.06	26.3	44.9	190
SC3	10/30/1999	1130	31	6.6	12.5	1.58	74.0	48.3	na	33.5	147	448
SC3	10/30/1999	1315	983	31.9	12.0	.274	61.1	19.7	na	25.1	32.7	109
SC3	10/30/1999	1435	2,066	29.1	10.9	.437	77.2	22.6	na	28.4	33.2	130
SC3	10/30/1999	1510	845	11.5	10.2	.515	82.7	24.3	na	28.1	35.7	155
SC3	10/30/1999	1610	443	21	9.79	.541	90.8	25.0	na	30.2	37.1	180
SC4	03/17/2000	0410	350	--	12.1	.870	65.2	44.8	.12	30.0	76.2	307
SC4	03/17/2000	0425	281	--	11.7	.649	75.3	32.8	.10	28.5	46.3	217
SC4	03/17/2000	0525	329	--	10.7	.536	95.5	24.1	.06	32.4	29.2	156
SC4	03/17/2000	0725	96.5	--	11.2	.642	104	29.0	na	38.0	33.6	217
SC4	03/17/2000	1150	23.5	--	10.5	.892	116	37.6	na	35.0	42.7	272
						Boggy Creek						
BC1	10/15/2000	2105	1,762	8.6	8.21	.470	48.7	25.8	.06	22.2	48.0	119
BC1	10/15/2000	2125	2,499	4.5	8.41	.445	50.0	19.2	.04	22.5	47.8	118
BC1	10/15/2000	2155	2,495	18.3	8.23	.502	50.4	19.7	.05	22.8	51.7	126
BC1	10/15/2000	2240	1,158	5.0	8.69	.496	52.6	20.1	.05	24.7	50.7	129
BC1	10/16/2000	0110	444	2.1	9.61	.537	56.1	20.5	.07	33.3	51.0	136
BC2	01/10/2001	0940–1015	87.5	2.1	16.4	1.11	61.3	22.3	na	24.8	50.1	223
BC2	01/10/2001	1040–1130	430	3.4	10.9	.641	53.5	21.1	.05	22.3	42.5	168
BC2	01/10/2001	1215–1245	522.5	7.9	9.68	2.84	50.9	23.0	.05	21.2	42.6	164
BC2	01/10/2001	1300–1340	469.5	4.0	10.9	.711	54.4	21.8	.05	21.5	44.3	164
BC2	01/10/2001	1400–1450	243.5	2.1	10.6	1.01	54.1	23.9	.06	21.6	53.4	198

Appendix 6.2. Organochlorine compound concentrations associated with suspended sediment in discrete rain event samples at Shoal and Boggy Creeks.

[In micrograms per kilogram. PCB, polychlorinated biphenyls; <, less than; E, estimated; --, not analyzed]

Rain event	Date	Time	Dieldrin	Chlordane	DDD	DDE	DDT	Total DDT	PCB Aroclor 1242	PCB Aroclor 1254	PCB Aroclor 1260	Total PCBs
						Shoal Creek						
SC1	03/12/1999	1200	<4.8	E36	8.0	10	7.6	26	<48	<48	<48	<145
SC1	03/18/1999	2120	2.2	16	1.4	7.9	4.1	13	<9.4	--	<9.4	<28
SC1	03/18/1999	2147	1.5	E11	E1.0	3.7	2.2	7.0	<11	<11	<11	<34
SC1	03/18/1999	2230	1.0	9.0	1.6	2.3	2.3	6.2	<8.5	<8.5	<8.5	<26
SC1	03/19/1999	0144	<8.6	<83	E6.0	14	8.6	28	<29	<29	<29	<87
SC2	04/26/1999	0910	34	339	<1.7	1.7	<1.7	1.7	<17	<17	<17	<51
SC2	04/26/1999	1145	9.9	40	<4.0	9.9	20	30	<40	<40	<20	<60
SC2	04/26/1999	1205	2.8	11	<1.1	4.4	11	15	<11	<5.5	<5.5	<22
SC2	04/26/1999	1247	5.1	<26	<2.6	5.1	7.7	13	<28	<28	<28	<84
SC2	04/26/1999	1347	<6.2	<62	<6.2	9.4	<10	9.4	<51	<51	<51	<154
SC3	10/30/1999	1130	28	<119	<12	32	45	--	<119	<119	<119	<358
SC3	10/30/1999	1315	.92	<8.7	<.87	2.0	1.8	--	<8.7	<8.7	<8.7	<26
SC3	10/30/1999	1435	<1.7	<17	<1.7	2.1	E1.5	--	<17	<17	<17	<50
SC3	10/30/1999	1510	<1.6	<16	<1.6	2.9	<1.6	--	<16	<16	<16	<47
SC3	10/30/1999	1610	<1.4	<14	<1.4	<1.4	<1.4	--	<14	<14	<14	<43
SC4	03/17/2000	0410	27	629	6.6	18	63	87	<16	25	E13	E37
SC4	03/17/2000	0425	8.9	85	2.6	16	36	54	<25	E17	E10	E27
SC4	03/17/2000	0525	<2.5	<25	E1.5	8.5	10	20	<25	E7.9	E7.9	E16
SC4	03/17/2000	0725	<2.8	<28	2.8	6.1	9.2	15	<28	<28	<28	<80
SC4	03/17/2000	1150	<4.3	<45	4.3	5.1	23	28	<43	<43	<43	<221
						Boggy Creek						
BC1	10/15/2000	2105	<4.5	74	E62	230	E120	E412	<35	E30	E34	E64
BC1	10/15/2000	2125	<3.7	43	E26	73	70	E169	<25	E19	47	E66
BC1	10/15/2000	2155	5.2	39	E15	E46	43	E104	<20	E10	E12	E22
BC1	10/15/2000	2240	5.6	55	E21	E72	75	E168	<55	E14	E15	E29
BC1	10/16/2000	0110	<8.0	E45	E20	73	76	E169	<80	E16	E28	E44
BC2	01/10/2001	0940–1015	<56	<555	<56	57	<74	57	<555	<555	<555	<1,665
BC2	01/10/2001	1040–1130	<6.0	E35	<6.0	43	<44	43	<60	E14	E12	E26
BC2	01/10/2001	1215–1245	<4.0	52	E4.2	58	E66	E128	<40	E15	E13	E28
BC2	01/10/2001	1300–1340	<2.0	46	E3.4	40	E41	E84	<20	E10	E10	E20
BC2	01/10/2001	1400–1450	<7.0	E72	<7.4	<7.0	<33	<ND	<70	<70	<70	<210

Appendix 6.3. Polycyclic aromatic hydrocarbon concentrations associated with suspended sediment in discrete rain event samples at Shoal and Boggy Creeks.

[In micrograms per kilogram. PAH, polycyclic aromatic hydrocarbons; $\Sigma\text{PAH}_{\text{SQG}}$, total PAHs used for consensus-based sediment-quality guidelines (Ingersoll and others, 2000); 2+3/comb, ratio of 2- and 3-ringed PAHs to combustion PAHs; E, estimated; <, less than]

Rain event	Date	Time	Total PAH	$\Sigma\text{PAH}_{\text{SQG}}$	Anthracene	Fluorene	Naphthalene	Phenanthrene	Benz[*a*]anthracene	Benzo[*a*]pyrene	Chrysene	Fluoranthene	Pyrene	2+3/comb
							Shoal Creek							
SC1	03/12/1999	1200	24,290	9,246	157	E56.6	67.5	928	699	627	1,807	2,651	1,687	0.13
SC1	03/18/1999	2120	39,248	8,733	E116	E40.9	E16.7	E911.7	E754	1,635	E1,415	2,012	E1,446	.05
SC1	03/18/1999	2147	31,595	15,459	163	E67.6	28.2	1,804	1,240	1,635	2,593	4,171	3,269	.10
SC1	03/18/1999	2230	13,070	5,760	56.2	E20.9	E9.4	659	488	452	1,160	1,709	1,026	.10
SC1	03/19/1999	0144	11,608	4,496	E45.7	E21.6	E9.5	543	440	388	1,034	750	1,034	.11
SC2	04/26/1999	0910	67,490	31,775	322	E105.1	142	3,220	2,203	3,220	5,763	8,475	7,119	.10
SC2	04/26/1999	1145	17,861	8,070	81.3	E29.8	E19.8	694	655	873	1,468	2,183	1,825	.10
SC2	04/26/1999	1205	4,138	1,695	17.7	E5.1	E4.1	138	138	199	331	436	359	.09
SC2	04/26/1999	1247	10,502	4,824	41.0	E14.4	E13.8	487	308	410	897	1,385	1,128	.10
SC2	04/26/1999	1347	18,123	7,598	E62.5	E22.9	E18.8	750	479	750	1,458	2,188	1,667	.10
SC3	10/30/1999	1130	55,629	21,594	235	E87.1	E93.5	1,548	1,484	2,806	3,871	6,129	4,516	.11
SC3	10/30/1999	1315	3,807	1,642	23.7	E8.2	E6.8	150	136	189	247	460	368	.13
SC3	10/30/1999	1435	8,306	3,530	35.5	E13.0	E13.0	379	213	343	556	1,065	805	.11
SC3	10/30/1999	1510	13,059	5,560	45.1	E15.3	E13.3	542	293	542	971	1,761	1,242	.10
SC3	10/30/1999	1610	3,000	1,186	E14.5	E4.3	E3.4	99.4	66.8	126	217	361	253	.11
SC4	03/17/2000	0410	34,453	17,090	223	65.7	E12.3	1,829	1,600	2,057	3,143	4,286	3,429	.11
SC4	03/17/2000	0425	33,001	14,560	178	E56.9	E30.2	1,566	1,139	1,886	2,811	3,559	2,918	.13
SC4	03/17/2000	0525	14,426	6,614	E75.99	E18.8	E8.2	699	426	699	1,185	1,884	1,398	.12
SC4	03/17/2000	0725	12,668	5,819	E58.0	E15.5	E12.4	653	352	549	1,005	1,762	1,244	.14
SC4	03/17/2000	1150	20,634	9,468	E97.9	E33.2	E42.6	1,106	468	809	1,830	2,851	2,000	.14
							Boggy Creek							
BC1	10/15/2000	2105	12,215	5,832	72.4	E21.4	E18.5	427	570	801	1,170	1,440	1,160	.16
BC1	10/15/2000	2125	8,530	4,184	43.6	E10.4	E9.3	286	409	583	836	1,060	849	.12
BC1	10/15/2000	2155	17,004	8,349	79.6	E22.5	E13.8	628	772	1,090	1,690	2,170	1,720	.10
BC1	10/15/2000	2240	13,253	6,647	69.1	E18.2	E14.1	482	633	900	1,340	1,710	1,340	.11
BC1	10/16/2000	0110	9,553	4,542	E53.3	E15.7	E13.4	349	415	623	978	1,120	836	.14
BC2	01/10/2001	0940–1015	12,595	4,012	E156	<570	E41.8	E280	E419	E560	E557	955	E734	.44
BC2	01/10/2001	1040–1130	29,479	10,207	138	E44.3	E31.5	851	887	1,340	1,720	2,810	2,050	.20
BC2	01/10/2001	1215–1245	34,106	10,996	136	E39.4	E25.7	848	1,020	1,400	1,850	3,060	2,310	.26
BC2	01/10/2001	1300–1340	23,970	10,203	139	38.6	E20.3	795	943	1,300	1,690	2,850	2,130	.16
BC2	01/10/2001	1400–1450	38,198	14,206	208	E57.1	E32.8	1,070	1,380	1,890	2,260	3,900	2,990	.13

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