shown in figure 8B. The exceptions were March and April 2000.

APPROACH

An integrated approach is needed to assess the effects of land use on water quality. Bed sediment, fish-tissue, aquatic-community structure, and water chemistry are environmental indicators that represent stream conditions at differing time scales. Water samples represent water quality at the time of sample collection. Benthic invertebrates can integrate water-quality conditions over a span of many weeks to year(s) and the fish-community structure can indicate stream conditions over many years (Wynn and others, 2001). Because of these varying time scales, it is important to interpret the sampling results for these water-quality indicators within the context of spatial and temporal scales of condition and response.

Authors of previous studies have concluded that both bed-sediment and fish-tissue samples are required for a complete assessment of the occurrence and distribution of trace elements (Wynn and others, 2001). Many constituents, such as trace elements and organic compounds, may be present in water but commonly are at concentrations that are difficult to quantify. These constituents are more likely to be detectable or even elevated in other sample media such as bed sediment, due to the tendency of some contaminants to adsorb onto small particles or bioaccumulate in the tissues of aquatic organisms.

Another important aspect is the condition of the hydrologic system at the time of sampling. Surface-water samples were collected over a wide range of flow conditions, including low flow and high flow, in an attempt to characterize the water quality during varying hydrological conditions. High flow was defined to include samples collected during an actual storm (rising limb and[or] peak of the hydrograph), as well as samples collected directly after a storm (on the falling limb of the hydrograph). If the sample was collected after the stream returned to pre-storm levels, then the sample was not considered to be a high-flow sample. Of the 14 high-flow samples, 2 were collected on the rising limb, 5 were collected at the peak of the hydrograph, and 7 were collected on the falling limb. Concentrations of contaminants from nonpoint sources typically increase during a storm as a result of overland runoff, but concentrations of contaminants from point sources may decrease during a storm as a result of dilution. By understanding the hydrologic condition at the time of sampling and reviewing the data in this context, the influence of point and nonpoint source contributions of contaminants on water quality can be examined.

This study was designed to assess the current conditions of Village and Valley Creeks by using three specific approaches: (1) water-quality, bed-sediment, fish-tissue, and aquatic-community structure data were collected from sites on Village and Valley Creeks and compared with data from two less-urbanized sites (LCR and FMC) to evaluate the effects of urban land use on water quality; (2) sites along Village and Valley Creeks were evaluated in an upstream-downstream order to assess which sites were most affected by specific constituents; and (3) water-quality data collected during low and high flow at each site were examined to assess the potential source(s) of contaminants.

Data-Collection Methods

The data-collection methods used during this study for water quality, bed sediment, fish tissue, aquatic communities, and stream habitat are described in this section. Field methods and laboratory methods are discussed.

Water-Quality Samples

Surface-water samples were collected during the period February 28, 2000, through February 14, 2001. Additional samples were collected at VIL-1, VAL-2, and FMC in May 2000 and at FMC in March 2001 as part of the USGS NAWQA Program. The frequency of sampling varied at different sites (table 5), primarily due to the modification of site selection in August 2000. Samples were not collected at FMC in October 2000 because the stream was dry. Each surface-water sample was analyzed for nutrients, major ions, total organic carbon (TOC), wastewater indicators, and fecal bacteria. Sampling for other constituents, such as trace elements, pesticides, PAHs, 5-day biochemical oxygen demand (BOD_5), and chlorophyll *a* and *b* was less frequent. Samples for BOD_5 and chlorophyll *a* and *b* analysis were shipped to the USGS Ocala Water Quality and Research Laboratory in Ocala, Florida. All other water samples were shipped to the USGS National Water Quality Laboratory (NWQL) in Denver, Colorado, for analysis.

Continuous water-quality monitors, installed at two sites on Village Creek (VIL-3 and VIL-4) before this study was initiated, provided a continuous record of water temperature, specific conductivity, and dissolved oxygen for the period between March 2000 and March 2001. In conjunction with this study, additional monitors were installed upstream from VIL-1 and at VAL-3, and

Table 5. Sampling type and frequency of water-quality and biological constituents at sites in the Birmingham area, Alabama, 2000–01

[NAWQA, National Water-Quality Assessment Program; BOD_5, 5-day biochemical oxygen demand; GC/MS, gas chromatography/mass spectrometry; HPLC, high-performance liquid chromatography; PCBs, polychlorinated biphenyls]

Site label (fig. 1)	NAWQA land-use gradient site	Type and frequency of water-quality sampling											
		Field properties[a], nutrients[b], major ions[c]	BOD_5[a]	Chlorophyll *a* and *b*[a]	Enterococci[a]	*Escherichia coli*[a]	Fecal coliform[a]	Wastewater indicators[d]	Total organic carbon[a]	Trace and major elements[e]	Pesticides (GC/MS)[f]	Pesticides (HPLC)[f]	Polycyclic aromatic hydrocarbons[g]
VIL-1	yes	12	5	6	9	12	11	10	10	6	6	6	4
VIL-2	no	7	6	7	7	7	7	7	7	4	3	3	3
VIL-3	no	10	7	7	8	9	9	10	10	5	3	4	3
VIL-4	no	3	0	0	2	3	3	3	3	2	1	1	1
VAL-1	no	10	6	6	8	10	10	10	10	6	5	5	6
VAL-2	yes	11	6	7	9	11	10	10	10	6	4	4	5
VAL-3	no	10	6	6	9	10	10	10	10	6	4	4	4
LCR	no	3	0	0	2	3	3	3	3	2	2	2	2
FMC	yes	8	7	7	6	8	7	6	5	4[h]	5	5	2

Site label (fig. 1)	NAWQA land-use gradient site	Type and frequency of biological sampling						
		Bed sediment trace and major elements[i]	Bed sediment pesticides and other organic compounds[j]	Fish (liver tissue) trace and major elements[k]	Fish (whole body) pesticides and PCBs[l]	Habitat survey[m]	Benthic invertebrate community survey[n]	Fish community survey[o]
VIL-1	yes	1	1	1	1	1	2	1
VIL-2	no	1	1	1	1	0	0	0
VIL-3	no	1	1	1	1	1	2	1
VIL-4	no	0	0	0	0	0	0	0
VAL-1	no	1	1	1	1	1	2	1
VAL-2	yes	1	1	1	1	1	2	1
VAL-3	no	0	0	0	0	0	0	0
LCR	no	0	0	0	0	1	2	1
FMC	yes	1	1	1	1	1	1	1

[a] See appendix table 2-2.
[b] See table 8.
[c] See appendix table 2-1.
[d] See appendix table 2-3.
[e] See table 15.
[f] See table 16 and appendix table 2-4.
[g] See appendix table 2-5.
[h] For mercury only, the sample size was 3.
[i] See appendix table 3-1.
[j] See table 22
[l] See table 24.
[m] See table 25.
[n] See appendix tables 3-2 and 3-3.
[o] See appendix table 3-4.

turbidity probes were added to the monitors at VIL-1, VIL-3, and VAL-3. All four measurements were recorded at VIL-1 and VIL-3 between April 2000 and February 2001; all four measurements were recorded at VAL-3 between June 2000 and February 2001.

Data-collection procedures, which conformed to standard USGS protocols (Wilde and others, 1999), included equal-width increment sampling (Shelton, 1994). Equal-width increment sampling produces a composite sample that is representative of flow in a cross section. Most water samples were collected by using a DH-81 sampler (Edwards and Glysson, 1999). Storm samples were not flow-weighted composite samples taken at specific intervals, as described in a U.S. Environmental Protection Agency (USEPA) sampling guide (U.S. Environmental Protection Agency, 1992b); instead, the storm samples were discrete. Field measurements of stream discharge, air temperature, water temperature, pH, dissolved oxygen, and specific conductance were made at the time of sampling.

A Teflon cone splitter and bottles were used to composite and split the water samples into separate sample bottles for various analyses. After splitting, water samples for dissolved nutrients and major ions were filtered by using a 0.45-micron (μm) pore size filter that was pre-rinsed with deionized water and native streamwater. Samples for dissolved pesticide analyses were filtered by using a 0.7-μm pore size glass-fiber filter. Wastewater indicator samples and PAH samples were collected as grab samples directly from the stream in 1-liter (L) glass bottles. Samples were preserved and chilled immediately after filtration and shipped overnight to the USGS laboratories in Denver and Ocala. Pesticide samples were analyzed by using gas chromatography/mass spectrometry (Zaugg and others, 1995) or high-performance liquid chromatography (Werner and others, 1996). All equipment that was used to collect and process samples was cleaned with a 0.2-percent nonphosphate detergent and rinsed with tap water and deionized water. Equipment was rinsed with a solution of 5-percent hydrochloric acid followed by deionized water if metals were sampled. A rinse of pesticide-grade methanol was added if organic compounds were sampled.

Water samples for analysis of fecal-indicator bacteria were processed in the field by USGS personnel for fecal coliform, *Escherichia coli* (*E. coli*), and enterococci (U.S. Environmental Protection Agency, 1997). Samples were collected by using an autoclaved 1-L polyethylene bottle with the DH-81 sampler. Samples were treated with a solution of 10-percent sodium thiosulfate to counteract the effects of residual chlorine in the water and processed within 6 hours of collection by membrane filtration techniques, as described in the USGS National Field Manual (Myers and Wilde, 1999). Results were reported as colonies per 100 milliliters (col/100 mL).

Bed-Sediment Samples

Bed-sediment samples were collected in October 2000 at VIL-1, VIL-2, VIL-3, VAL-1, VAL-2, and FMC to determine the concentration of trace and major elements. Sediment samples were collected from the upper 3 centimeters (cm) of fine sediment in depositional areas of each stream reach following the protocols described in Shelton and Capel (1994). Several samples were collected from each of five depositional areas, composited, and homogenized by mixing. The composited material was processed through a 0.063-millimeter (mm) mesh nylon screen. Samples were sent to the NWQL for analyses of trace and major elements.

The NWQL analytical procedure for trace and major elements in bed sediment uses multi-acid digestion and inductively coupled plasma-mass spectrometry techniques. Results are reported in micrograms of analyte per gram dry weight of sediment, or as a percentage of dry weight (Briggs and Meier, 1999). This method provides a total extractable metals concentration that includes mineral-bound metals. Nine major elements—including aluminum, calcium, iron, magnesium, phosphorus, potassium, sodium, sulfur, and titanium—were reported as a percentage of dry weight. Concentrations of total carbon, organic carbon, inorganic carbon, and 36 trace elements were reported in concentrations as microgram(s) per gram (μg/g) of bed sediment (dry weight). Trace elements are defined as elements that usually occur in concentrations less than 1,000 μg/g (Forstner and Wittmann, 1979). Ten trace elements—arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc—are classified as priority pollutants (Code of Federal Regulations, 1996) because in low concentrations they are toxic to aquatic organisms; however, some of these trace elements are necessary for metabolic processes in aquatic organisms.

A subsample of the composited bed-sediment material, collected before the remainder was processed through the screen, was analyzed for particle-size composition at the USGS Cascades Volcano Observatory Laboratory in Vancouver, Washington, by using a sand-only procedure (Guy, 1969). The sand-only analysis was used to determine the percentage of bed-sediment material that was less than 0.063 mm in diameter. Particulate size is important in understanding the concentration and distribution of trace element in the

environment because trace elements tend to adsorb onto the fine particulates of bed sediment (Horowitz, 1991).

Sediment samples from six sites were analyzed for the presence of organochlorine pesticides, chlorpyrifos, six other organophosphate pesticides, and other organic compounds. Initial sample collection for the analysis of pesticides and other organic compounds was the same as for the trace elements. However, aliquots of the homogenized mixture were processed through a 2-mm mesh stainless steel sieve, placed in a methanol-washed glass container, and preserved on dry ice for transport to the NWQL. The NWQL analytical procedure for organics in bed sediment includes dual capillary-column gas chromatography with electron-capture detection. Results are reported in microgram(s) of analyte per kilogram of the wet weight of the sediment (Foreman and others, 1995).

Fish-Tissue Samples

Fishes were collected in October 2000 at six sites coincident with bed-sediment sample collections. The fishes were collected by use of a backpack mounted, DC-powered electrofishing unit following NAWQA protocols (Meador and others, 1993). The targeted fish species was the longear sunfish (*Lepomis megalotis*) because of its reported abundance and feeding habits. However, fish collections at the FMC site failed to yield enough longear sunfish to provide adequate liver tissue. Therefore, bluegill sunfish (*Lepomis macrochirus*) also were collected. The longear sunfish was not captured at VIL-1, so the bluegill sunfish was used for tissue assessment at that site. Fish-tissue samples were taken from the targeted species and processed in accordance with NAWQA protocols (Crawford and Luoma, 1993).

Fishes were processed on site for two types of tissues, liver and whole-body tissue minus the livers. Tissue samples from the two species of fishes collected from FMC were composited and submitted to NWQL as a single sample. For analysis of trace-element concentrations, the livers of at least five individuals from each site were removed by use of a ceramic knife and Teflon forceps, weighed, placed in acid-cleaned glass containers, preserved with dry ice, and shipped overnight to the NWQL. In the laboratory, tissue samples were processed by nitric acid and hydrogen peroxide digestion. Acid-processed samples were dried, reconstituted with a 5-percent nitric acid solution, and filtered. The filtrate was diluted to a specific volume. The analysis of fish-liver tissue for concentrations of trace elements was conducted by using inductively coupled plasma/mass spectrometry or inductively coupled plasma/atomic emission spectroscopy for all trace elements except mercury. Procedures for mercury analysis incorporate cold vapor atomic absorption spectrophotometry (Olson and DeWild, 1999). All results were reported as dry weight, total recoverable concentrations in microgram(s) of analyte per gram of tissue, as detailed by Hoffman (1996).

For the analyses of organochlorine pesticides and polychlorinated biphenyls (PCBs), the remainder of each fish (whole body minus the liver) was wrapped in aluminum foil and preserved with dry ice. The fish tissues were shipped overnight to the NWQL and analyzed for organochlorine pesticides by capillary-column gas chromatography with electron-capture detection (Leiker and others, 1995). Successful detection of organic compounds and the levels of detection can vary from site to site because of the inherent variability of biological tissues. Results are reported as microgram(s) of analyte per kilogram of tissue (µg/kg), wet weight. Values reported with a "less than" symbol (<) are considered to be nondetections. Values reported with an "E" are considered to be estimates because definitive quantification was not possible. Estimated values are considered to approximate actual values for the purpose of comparative evaluations and statistical analysis.

Aquatic-Community Samples

Benthic invertebrates and fishes were collected following procedures outlined in Cuffney and others (1993) and Meador and others (1993), respectively. The health of these organisms is often directly related to changes in water quality and habitat. Changes in the composition of an aquatic community and functional changes in the ecosystem can result from exposure to contaminants, changes in the riparian zone, or changes in the hydrology of the aquatic system. For example, changes in the numbers and types of algae and aquatic plants due to exposure to herbicides or nutrients, may result in changes in the numbers and types of aquatic organisms that use them for shelter and resources. Benthic invertebrates occupy diverse functional niches in aquatic ecosystems. They recycle organic matter, consume smaller organisms, and are important components in the diet of fishes. Benthic invertebrates are commonly used to assess the health of aquatic communities because they are easy to collect and identify, usually abundant, and relatively sessile (Merritt and Cummins, 1996).

Benthic-invertebrate samples were collected at VIL-1, VIL-3, VAL-1, VAL-2, FMC, and LCR. One collection was made at each of six sites in June 2001. A second collection was made at the same sites in October 2000, except at FMC, where the stream was dry. All samples of benthic invertebrates were sent to the NWQL

for taxonomic evaluation and determination of benthic-invertebrate density, as described in Moulton and others (2000).

Quantitative samples of benthic invertebrates were collected from five riffle habitats at each site by using a 0.25 square meter (m^2) Slack sampler with a 425-µm mesh net (Cuffney and others, 1993) and then composited into one sample (1.25 m^2 total area). The quantitative collection provided an estimate of aquatic invertebrate richness and density in the targeted habitat. Qualitative samples were collected from all accessible habitat types at each site by using a D-frame net (210-µm mesh) and by hand-picking invertebrates from rocks and other substrates. The qualitative collection further characterizes the invertebrate taxa present throughout the sampling reach.

Fishes were collected in April and May 2001 at the same six sites. The primary fish-collection device was a backpack-mounted, DC-powered electrofishing unit. Two passes were made along the stream reach. Stunned fishes were netted and placed in a collection bucket. At the completion of the first pass, the collected fishes were identified, weighed, measured (standard length and total length), and evaluated for anomalies such as lesions, tumors, parasites, and eroded fins (Meador and others, 1993). Once processing was completed, the fishes were placed in a holding container to prevent them from returning upstream and being recaptured during the second pass. The second pass was made along the length of the reach and the fishes were processed in a like manner. Additional collections of fishes were made with a seine at VIL-1, VIL-3, VAL-2, and LCR to capture species that might have eluded the shocking effort. Fishes captured by seining were processed in the same manner as those captured by electrofishing. Seine collections at VAL-1 were impractical because of shallow depth and obstructions.

Fishes that could not be readily identified at each site were preserved in 10-percent buffered formalin and were sent to the USGS laboratory at the Florida Caribbean Science Center in Gainesville, Florida, for identification. All remaining fishes were released unharmed to the stream when processing was completed.

Stream-Habitat Characterization

Measures of habitat conditions and structure are important components of any ecological study. Several measures of biological condition are related to stream habitat (Fitzpatrick and others, 1998). For example, removal of trees from the riparian zone during construction may cause increased amounts of solar radiation to reach a stream's surface. Increased sunlight may lead to an increase in the number of photosynthetic organisms, such as algae and plants, which can influence the density of organisms that use them for food and shelter. The amount of infrared radiation that reaches a stream's surface can have a direct influence on the invertebrate community. Invertebrate emergence periods are often regulated by water temperature, and the effects of temperature influence the distribution patterns of aquatic insects (Ward, 1992).

Aquatic insects are closely associated with the bed material of the stream in which they live, at least for a portion of their lives. Bed substrate provides food, shelter, and habitat space. Therefore, the type of substrate in a stream influences the abundance and distribution of aquatic insects (Minshall, 1984). The size of substrate particles, the amount of organic material in and on the substrate, and the stability and texture of the substrate have been found to be of ecological importance (Ward, 1992; Allen, 1995).

Habitat assessments were made at six sites (VIL-1, VIL-3, VAL-1, VAL-2, LCR, and FMC) following the protocols described in Fitzpatrick and others (1998). Stream reaches ranging in length from 150 to 349 m were sampled. Within each reach, habitat characteristics were measured at 11 transects. A transect is an imaginary line across the stream, oriented perpendicular to stream flow. The first and last transect defined the start and end, respectively, of the stream reach. Transects were established at approximately equidistant intervals along each reach, and the habitat characteristics of at least seven points along each transect were evaluated. The points included the left and right edges of the water, three or more sites in the stream that corresponded to the thalweg (the deepest part of the channel) and one location on each side of the thalweg, and one or more points on each bank.

Three collection points in the wetted channel were made along each of the 11 transects resulting in a total of 33 collection points. Observations and measurements within the reaches and along the transects included many physical characteristics of the stream channel, quantitative evaluation of riparian-zone shading, and the amount and type of geomorphic channel units (runs, pools, riffles) in each stream reach (Fitzpatrick and others, 1998).

Data Analysis and Review

This section includes data analysis and review for water quality, bed sediment, fish tissue, aquatic communities, and stream habitat in the study area. Specific methods used to interpret data, including graphical and statistical presentation, are discussed.

Water-Quality Data

Methods used to interpret water-quality results in this report include various graphical tools and statistical methods. Graphical tools include the use of bar charts, which illustrate the speciation of certain nutrients (nitrogen and phosphorus) and the frequency of detection for other constituents (trace and major elements, wastewater indicators, pesticides). Box plots are used to display the variability in nutrient concentrations, and high-flow/low-flow figures are used to illustrate the concentrations of different constituents, as well as the hydrologic condition at the time of sampling. Only detected values (including estimated concentrations) are shown on the high-flow/low-flow figures—non-detections are not shown. If the concentrations detected during low flow were consistently higher than those during high flow, a "P" was placed on the graph, indicating point sources. If the concentrations detected during high flow were consistently higher than those during low flow, an "NP" was placed on the graph, indicating nonpoint sources. If the results were mixed, a "B" was placed on the graph, indicating that both point and nonpoint sources may be contributing. No symbols were placed on sites if this designation could not be made from the available data, such as at LCR, where high-flow samples were not collected. Data also were examined in terms of maximum concentrations and(or) ranges of concentrations, with respect to flow. Statistical methods could not be applied to evaluate the relation between discharge and concentration because the samples were collected over such a limited range of discharge (either high flow or low flow). This interpretation of the data, using high-flow/low-flow figures, can be useful in defining the influence of point and nonpoint source contributions of contaminants on water quality; however, the interpretation is limited because of the small sample size, and results should be viewed as preliminary or exploratory rather than conclusive.

The USGS NWQL has implemented new procedures for interpreting and reporting low-concentration data in water-quality samples (Childress and others, 1999). Concentrations of analytes that either were not detected or were not identified are reported as "less than" the laboratory reporting level (<LRL) and are considered to be nondetections. Analytes that were detectable at concentrations between the LRL and the long-term method detection level (LT-MDL), which is usually one-half the LRL, and that pass identification criteria were estimated. Estimated concentrations are noted with the remark code "E". The uncertainty associated with the magnitude of estimated concentrations is greater than that associated with values that were not estimated (Martin and others, 1999). The sample matrix and the instrument condition sometimes limit the reliable measurement of an analyte in the laboratory. The minimum reporting level (MRL) and(or) LRL for organic compounds have been calculated by the NWQL. The NWQL collects quality-control data on a continuing basis to determine the MRLs, LT-MDLs, and LRLs. These values are re-evaluated each year and, consequently, may change from year to year. Values listed in this report were those in effect on October 1, 2000.

Sensitive analytical methods used in this study resulted in low detection limits and higher frequencies for many pesticides. Comparison of detection frequencies among pesticides can be misleading because of the different LRLs associated with each of the pesticides. For example, atrazine has an LRL of 0.007 micrograms per liter (µg/L) and may have been detected more often than prometon, which has an LRL of 0.015 µg/L, even though prometon may have been present at significantly higher concentrations than atrazine. To reduce this type of bias when calculating detection frequencies, pesticide data were adjusted by censoring to a common threshold of 0.01 µg/L so that values less than 0.01 µg/L were not considered detections. These adjusted procedures were used when comparing the pesticide detection frequency between national data from the NAWQA Program and data from this investigation as part of the Birmingham Watersheds Project. Non-adjusted data were used when evaluating the frequency of detection for pesticides in the Birmingham area.

Median concentrations of constituents were used when comparing constituent levels between sites along the urban streams. Median concentrations represent the 50^{th} percentile of the concentration data and are less affected than mean concentrations by the value of extremely high or low concentrations. Median concentrations were not computed at two sites (LCR and VIL-4) because of the limited number of samples (three) at each site and the brief time that the samples were collected (February 28–July 1, 2000).

Nonparametric hypothesis tests were used to evaluate relations between water-quality parameters and land-use characteristics. The Spearman-rho rank sum correlation test was used to assess the strength of these relations (SAS Institute Inc., 1989). In this non-parametric test, data are represented by ranks rather than actual values. Median values of selected water-quality constituents were calculated and then compared to land-use characteristics. Logarithmic probability regression was used to predict the values of data below the detection limit prior to calculating median values. In many instances, median values at particular sites could not be

calculated due to either the large number of non-detections, multiple-detection levels, or the limited sample size. Correlation coefficients were calculated only for those parameters with high detection rates (greater than 50 percent). Correlation coefficients were examined only when median values could be determined at a minimum of five sites. Although statistically significant differences were found, the significance of the results and the power of the tests used are limited because of the small sample size. The results should be viewed as preliminary or exploratory rather than conclusive.

Correlation tests calculate a probability statistic (p) and a correlation coefficient (rho). The probability statistic relates to the confidence level. A probability statistic of 0.05, as used in this report, means that there is a 95-percent probability that the correlation is statistically significant. The correlation coefficient can range from -1 to +1 and describes the strength of the correlation and how the correlated parameters vary. The correlation coefficient, rho, is positive when one variable increases with the other and negative when one variable increases as the other decreases. For this report, significant correlation was determined by an absolute rho value of 0.7 or greater, provided that the p value was less than or equal to 0.05. All data sets with rho and p values within the designated ranges were verified by scatter plots to determine the distribution of the data. Plots that indicated poor distribution by showing grouped data points or outliers were not considered in the correlation analysis. The determination that a correlation existed meant that the data sets varied with each other in a constant pattern, but did not necessarily indicate a cause and effect relation (Helsel and Hirsch, 1995).

The Kruskal-Wallis test and the Tukey multiple-comparison test were used to test whether water-quality constituent concentrations at one site were significantly different from constituent concentrations at other sites (SAS Institute Inc., 1989). The Kruskal-Wallis test is a one-way nonparametric analysis of variance (ANOVA) that was used to determine whether significant differences existed between independent data groups—sites on Village Creek, sites on Valley Creek, and the combined sites (VIL-1, VIL-2, VIL-3, VAL-1, VAL-2, VAL-3, and FMC). The Tukey multiple-comparison test was then used to compare the differences in concentrations in an upstream-downstream order on each stream and to compare the concentrations of selected constituents in Village and Valley Creeks to concentrations at FMC. The simplest procedures for performing nonparametric multiple comparisons are rank transformation tests (Helsel and Hirsch, 1995). Ranks were substituted for the original data and the Tukey multiple-comparison test was performed on the ranks. Data were censored to the highest detection level whenever multiple detection levels were present. Statistical tests were not performed on parameters if censoring resulted in a severe (near 50 percent or more) loss of data (Helsel and Hirsch, 1995). As stated earlier, the significance of the results and the power of the tests used are limited because of the small sample size and the inherent limitations of statistical tests performed on small data sets. The results should be viewed as preliminary or exploratory rather than conclusive.

The USEPA has water-quality standards and guidelines for certain chemicals that can have adverse effects on human health, aquatic organisms, and wildlife. Although the Maximum Contaminant Levels (MCL) established by the USEPA pertain to finished drinking water supplied by a community water supply, they provide values with which the sampled concentrations can be compared (U.S. Environmental Protection Agency, 2001). Aquatic life criteria established by the USEPA and ADEM provide for the protection of aquatic organisms for short-term (acute) and long-term (chronic) exposures. In some instances, Canadian guidelines were used for comparisons when other criteria were not available. Fecal-bacteria concentrations were compared to established State and Federal standards and criteria. The USEPA defines criteria for single sample densities for *E. coli* and enterococci based on the designated use of the water (U.S. Environmental Protection Agency, 1986). The ADEM defines criteria for fecal coliform based on water-use classification, single sample density, and the geometric mean of at least five samples taken over a 30-day period (Alabama Department of Environmental Management, 2000d). Exceedance frequencies were calculated by summing the number of exceedances and dividing by the total number of samples collected for each type of bacteria.

Concentrations of total nitrogen, total phosphorus, and chlorophyll *a* in the study basins were compared to recommended criteria developed for Nutrient Ecoregion XI (U.S. Environmental Protection Agency, 2000a). The USEPA has identified quantified endpoints for these variables to provide for the protection and propagation of aquatic life and recreation, and to provide sufficient protection of uses (and to maintain downstream uses) on rivers and streams. Instantaneous loads, in kilograms per day, were calculated for total nitrogen and total phosphorus and were computed as the product of discharge and concentration at the time of sampling. Instantaneous yields, in kilograms per hectare per year, were computed as the product of the instantaneous load and the numbers of days in the year divided by the

drainage area in hectares. Accurate estimates of daily or annual loads could not be computed from the limited amount of data available.

The following equation was used to calculate instantaneous load:

$$L_i = Q_i \times C_i \times K \ . \quad (1)$$

The following equation was used to calculate instantaneous yield:

$$Y_i = \frac{L_i \times T}{(J \times DA)}, \quad (2)$$

where:

L_i = Instantaneous load in kilograms per day (kg/d) based on the discharge and concentration at the time of sampling;

Y_i = Instantaneous yield in kilograms per hectare per year (kg/ha)/yr;

Q_i = Instantaneous discharge in ft^3/s;

K = 2.447, correction factor for unit conversion from $(ft^3–mg)/(sec–L)$ to kg/d;

C_i = Instantaneous concentration in milligrams per liter (mg/L);

T = 365.25, the number of days per year;

DA = Drainage area, mi^2; and

J = 259, correction factor for unit conversion from square miles to hectare.

Quality-Control Methods and Results

Quality-assurance and quality-control measures were practiced throughout the study according to established USGS guidelines (Mueller and others, 1997). Laboratory and field blank samples were processed using water certified to contain undetectable concentrations of constituents to be analyzed. Data from blank samples were used to determine the extent of contamination potentially introduced during sampling, sample processing, shipping, or laboratory analysis. Blank water used for the inorganic constituent sample was distilled, deionized water obtained from the Ocala Water Quality Research Laboratory in Ocala, Florida. Blank water used for the organic constituent sample was either pesticide-grade or volatile-organic-compound-grade blank water obtained from the NWQL.

Four blank samples were analyzed for nutrients, major ions, organic carbon, and trace metals. No constituents were detected at levels greater than the LRL. Constituents that were detected in field blanks at levels below the LRL include dissolved nitrogen (ammonia + organic), total organic carbon, dissolved phosphorus, silica, copper, and chloride (appendix table 1-1). Constituents that were detected in the equipment blank at levels below the LRL include calcium, magnesium, and silica—zinc was detected at a level exceeding the low-level MRL for the equipment blank (appendix table 1-1). Six out of 75 environmental samples had nutrient concentrations at or near the levels found in the field blanks. One out of 41 environmental samples had copper concentrations at or near the level found in the field blanks. The concentrations of silica, chloride, calcium, magnesium, zinc, and total organic carbon found in the field and equipment blanks were substantially lower than concentrations found in stream samples. Three additional blank samples were analyzed for PAHs and pesticides—no constituents were detected in these blanks. These low-level detections indicate little potential for contamination of streamwater samples.

The method designed by the NWQL for wastewater indicators was considered experimental during this sampling period. Data were censored according to the detection level of constituents found in laboratory and field blanks. If a constituent were found in either a laboratory blank or a field blank and also detected in a stream sample during that same sampling trip at the same magnitude, then the detection was not included. The NWQL analyzed 18 laboratory blanks on batches including samples from this study. Eight of the 16 constituents that were examined in detail for this report were detected in laboratory blanks (appendix table 1-1). Seven field blanks were sent to the NWQL. Triclosan was the only one of the 16 constituents to be detected in a field blank (appendix table 1-1); triclosan was also detected in the laboratory blank for that particular batch of samples.

Sample replicates were collected to quantify the reproducibility of the results. Data from replicate samples were used to assess variability due to sample processing and laboratory analysis. The relative percentage difference (relative percentage difference = $|A - B| / [(A + B)/2]$) between the environmental samples and the corresponding replicate samples ranged from 0 to 54.6 percent with a median of 2.8 percent. Replicate results from nutrients, major ions, and metals indicated good reproducibility of data (less than 10-percent difference) in 95 percent of the detections (appendix table 1-2). Replicate results from pesticides indicated good reproducibility of data in 73 percent of the detections. For some pesticides, such as diazinon and simazine, the relative percentage difference (9.3–13.6 percent) was consistently higher than other pesticides, such as atrazine or prometon (2.0–4.0 percent;

appendix table 1-2). Replicate results from wastewater indicators showed good reproducibility of the data in 50 percent of the detections (appendix table 1-2).

The effectiveness of sterilization and processing procedures for bacterial analyses was checked by processing a sterile water blank at each site. Additional procedures included the regular analysis of procedure blanks as well as the frequent analysis of replicate samples.

Bed-Sediment Data

Sediment samples with particles larger in size than 0.063 mm have lower concentrations of trace elements because there is less surface area available for adsorption. A grain size ≤0.063 mm was selected for analysis for trace elements in bed sediment because of the underlying assumption that most, if not all, of the trace elements would be contained within that fraction (Horowitz, 1991). Concentrations of trace elements detected in the ≤0.063-mm fraction of the bed sediment may not be biologically available because of the strong attraction of the elements to particulates of this size. Therefore, particulate-bound concentrations of trace elements that exceed a known toxic limit may not present a toxic hazard. Changes in water chemistry (such as a decrease in pH), however, can facilitate the release of sediment-bound trace elements. Biological activity, such as the methylation of metallic mercury by microorganisms, also can remove elements from bed-sediment particulates and make them biologically available. Resuspension in the water column and transport of sediment-bound elements during storms and floods to locations downstream also may occur. Reservoirs, small impoundments, and backwaters can become sinks for trace elements, which can increase in concentration and toxicity as sediment is accumulated.

To determine whether a trace-element concentration may have an adverse effect on aquatic biota, it is useful to compare the concentration with a known toxic-effect level. Sediment-quality guidelines for the protection of aquatic life commonly are used for determining the potential toxicity of bed-sediment trace elements to aquatic organisms (Canadian Council of Ministers of the Environment, 1995). The probable effect level (PEL) is the concentration of an element or compound that is likely to cause an adverse effect on aquatic biota. The concentrations of trace elements in bed sediments were compared with the PELs and also with median concentrations determined from bed-sediment samples collected at sites across the Nation as part of the NAWQA Program (Rice, 1999).

Fish-Tissue Data

Aquatic organisms can accumulate trace elements and organic compounds in their bodies. This bioaccumulation can provide useful evidence about the occurrence and distribution of these substances. Concentrations of organic compounds and trace elements in fish tissue can be biomagnified to concentrations that are higher than those in the surrounding water or bed sediment (Laws, 1993; Brigham and others, 1998). It is important to quantify the concentration of specific substances in fish tissue because, while they may be detected in very low levels in the environment and thus be considered harmless, biomagnification can yield concentrations that may result in detrimental effects not only to biota that contain them but to organisms that consume them. Many trace elements are deposited in the liver, which is the primary detoxifying organ of the body. Organochlorine pesticides, such as DDT (1,1,1-trichloro-2,2-bis(p-chlorophenyl)ethane), are lipophilic and are stored in fat tissue.

In general, standards and guidelines aimed at the protection of human health apply to contaminant concentrations in the edible portion of the fish. The only national guidelines that apply to whole fish tissue are the preliminary recommendations made by National Academy of Science/National Academy of Engineering (NAS/NAE) in 1972, and these are aimed at the protection of fish-eating wildlife (National Academy of Science/National Academy of Engineering, 1973). Most standards and guidelines for pesticides in fish tissue apply to edible portions of fish rather than whole fish—and contaminant-residue data in whole fish cannot be compared directly with them—except as a screening procedure to determine whether additional sampling is warranted (Nowell and Resek, 1994). NAS/NAE and Canadian standards for trace-element concentrations in fish-liver tissue do not exist. The concentrations of trace elements, organochlorine pesticides, and PCBs detected in fish tissue were compared among sites and to the available standards and guidelines.

Aquatic-Community Data

Benthic-invertebrate community data were processed by using the Invertebrate Data Analysis System (IDAS), a computer program developed for evaluating invertebrate communities for the NAWQA Program (T.F. Cuffney, U.S. Geological Survey, written commun., October 2001). The output of the program includes diversity and similarity indices that allow comparisons of community attributes. For this study, two analyses were conducted. The first was based on the results of the

quantitative data collected in each stream, and provided information on invertebrate density and community composition. The second analysis included the combined results of the quantitative and qualitative invertebrate data and, using IDAS, provided species-richness information and calculated an index of similarity. The invertebrate collections were divided into three groups: (1) insects without midges (Chironomidae), (2) midges, and (3) non-insect invertebrates. Results of the community assessments were used to evaluate possible relations with bed sediment, habitat, and land use for each sample site.

Community metrics based on benthic-invertebrate structure may be indicative of water quality. For streams in the Birmingham area, the metrics were chosen from a subset of rapid bioassessment protocols used by the USEPA (Plafkin and others, 1989). These metrics include:

Community richness – This metric is a measure of the number of taxa present in the community. In general, the number of taxa decreases as the water quality decreases. The fewer taxa, the more likely the community has been degraded.

Diversity – The Shannon index of diversity (sometimes called the Shannon-Wiener index) was used to evaluate the diversity of the benthic-invertebrate community at the study sites. This index is based on the proportional abundance of species and accounts for both species richness and evenness (Magurran, 1988).

Density – This measurement refers to the total number of individual organisms within a specified area, such as a square meter. Density can be calculated for the entire community or for individual species or trophic levels. The density of the benthic-invertebrate community may be indicative of changes in habitat or water quality in a stream. Under stressful conditions, sensitive organisms disappear and tolerant organisms increase in numbers. Communities with only a few dominant species generally are considered to be stressed.

Similarity – Similarity is a measure of how alike two communities are. The use of similarity indices is based on the assumption that communities become more dissimilar as stress increases (Rosenberg and Resh, 1993). The measure is a comparison of the taxonomic structure of a representative sample of the community at a selected site and time with that of an index site. The higher the value of the similarity index, the more similar the two communities. A Pinkham-Pearson similarity index was calculated for the benthic-invertebrate communities sampled in streams in the Birmingham area, comparing the benthic-invertebrate community in each site to that in FMC. This index measures the degree of similarity in communities between sites, incorporating presence/absence data, abundance, and the types of taxa present.

The Ephemeroptera, Plecoptera, Trichoptera (EPT) index – This index refers to the number of mayfly (Ephemeroptera), stonefly (Plecoptera), and caddisfly (Trichoptera) taxa in a sample. Total EPT richness is an important indicator of water quality because these insects are known to be relatively sensitive to contamination. The EPT index has been empirically shown to track other indicators of ecological degradation (Wallace and others, 1996).

The ratio of EPT abundance to chironomid abundance – This ratio can be indicative of changing water quality. As the ratio decreases in magnitude, the proportion of EPT taxa decreases. A disproportionate number of chironomids relative to the EPT taxa may be indicative of environmental stress.

Relative abundance – The relative abundance of a taxon is the proportional abundance of that taxon within the sample or community; that is, the percentage of the total number of individuals in the community that is represented by that taxon. Relative abundance can indicate whether one or more taxa comprise an unusually large percentage of the community. If there is an equitable distribution of individuals within all of the taxa, the community appears to be well balanced with no taxon being unusually dominant. In general, when the number of individuals in a few taxa or a single taxon is disproportionately greater than that of any other taxon of the same type, it is likely there has been an environmental perturbation.

Community metrics (measures of community structure and function) have been developed to evaluate the relative health of fish communities and can be used as an indicator of environmental stress (Plafkin and others, 1989). Metrics used in this study include: (1) total number of fish species, (2) community diversity, (3) similarity of fish communities among sample sites, (4) the number and identification of darter and sculpin species, (5) the relative abundance and identity of sunfish species, (6) the number and identity of minnow or sucker species, (7) the relative abundance of green sunfishes, (8) the number of intolerant taxa, and (9) the relative abundance of anomalies among the fishes.

The sampling time for each reach differed, depending on the length and width of the reach and the complexity of habitat. To reduce the bias associated with different sampling times, the total electrofishing time (application of power to the water) was converted to units of effort. A unit of effort was defined as 300 seconds (5 minutes) of power application. Total numbers of fishes captured at each site were divided by the units of effort to

provide a more equitable comparison of fish capture among sites.

The Sorenson similarity index was calculated to determine how similar the fish communities were between the reference site (FMC) and the other sampling sites. This is a widely used index based on species presence and absence, and is designed to equal 1 in the case of complete similarity (Magurran, 1988):

$$S = \frac{2c}{(a+b)}, \quad (3)$$

where

S = Index of similarity,

a = the number of species occurring in the first site,

b = the number of species occurring in the second site, and

c = the number of species common to both sites.

Benthic-invertebrate data and fish-community data also were compared among the streams, with land-use and habitat characteristics, and with chemical and physical characteristics of the bed sediment by using the Spearman-rho rank correlation test (SAS Institute Inc., 1989).

Stream-Habitat Data

The habitat data evaluated in this report represent major components of habitat along each transect and in each stream reach. The habitat features evaluated include the physical characteristics of the stream channel, the water depth and wetted channel width, the amount of shading by riparian vegetation, the amount of ground cover along the banks, and the percentage of each type of geomorphic channel units (pools, runs, riffles) in the reaches. Habitat characteristics were compared among the streams and with benthic-invertebrate and fish-community data, land-use data, and with chemical and physical characteristics of the bed sediment by using the Spearman-rho rank correlation test (SAS Institute Inc., 1989).

RESULTS AND DISCUSSION

This section includes the analytical results and discussion related to water chemistry, bed-sediment and fish-tissue samples, and an evaluation of the benthic-invertebrate and fish communities in the study area.

Water Quality

This section includes the results and discussion of water-quality data evaluated for this study. Topics discussed include analytical results related to basic water chemistry, major ions, field and continuous measurements of water properties, nutrients, fecal indicator bacteria, wastewater indicators, trace elements, pesticides, and PAHs.

Basic Water Chemistry

The chemistry of surface water is the result of interactions between rain, ground water, rocks, and soils near the Earth's surface. Dissolved and particulate constituents enter a stream by surface runoff, precipitation, or ground-water discharge. The major dissolved constituents that give water its characteristic chemistry are cations and anions. Cations are positively charged and include calcium, magnesium, sodium, and potassium. Anions are negatively charged and include chloride, nitrate, sulfate, bicarbonate, and carbonate. The concentrations of these dissolved ions generally are reported in parts per million (milligrams per liter).

Streamwater chemistry varies with flow conditions because flow pathways change in the watershed. Under low-flow conditions, streamwater is predominantly ground-water discharge. The nature and concentration of dissolved constituents are dependent on the composition of the aquifers through which the ground water flows. During and immediately after a storm event, streamwater is a mixture of rainwater and surface runoff, shallow subsurface flow through the soil zone, and ground-water discharge. Precipitation produces an overall dilution of the major ion composition. Human activity also can alter water chemistry by contributing additional ions, such as sodium and chloride, from leaking or overflowing sewer systems, industrial discharge, or urban runoff. Although basic ions are not considered contaminants, elevated levels (above natural background levels) may indicate potential sources of contaminants (nutrients, trace elements, synthetic organic compounds).

Major Ions

Major ions constitute the greatest part of the dissolved solids in water. A summary of major ion concentrations, expressed in milliequivalents per liter (meq/L), during low, median, and high flow is presented in appendix table 2-1. Concentrations were summarized for all sites except VIL-4 and LCR (due to the small sample sizes). All sites exhibited similar water quality, characterized by a strong calcium-bicarbonate

component, which was most pronounced during low flow, due to ground-water discharge from the underlying carbonate rocks in the stream valleys. The bicarbonate component was most pronounced at VIL-1. The water chemistry at VIL-1 and FMC exhibited the lowest median concentrations of sodium, potassium, chloride, and sulfate. The similarity of the basic water chemistry at VIL-1 and FMC indicates that VIL-1 is the least impacted (of the Village and Valley Creek sites) from urbanization. The highest level of chloride was detected at VIL-2 during low flow. Median levels of chloride were highest at VIL-3, VAL-1, and VAL-2. Median levels of sodium were highest at VAL-1 and VAL-2, indicating that water chemistry at these sites may be more strongly influenced by anthropogenic factors. All sites exhibited the effects of dilution during storm events.

The Kruskal-Wallis test and the Tukey multiple-comparison test were applied to the major ion data at seven sites to determine if the variations in water chemistry among sites were statistically significant (table 6). Although statistically significant differences were found for several of the major ions, the significance of the results and the power of the tests used are limited because of the small sample sizes. Potassium and sulfate concentrations at VIL-2, VIL-3, VAL-1, VAL-2, and VAL-3 were significantly greater than concentrations at FMC. Chloride concentrations at VIL-2, VIL-3, VAL-1, and VAL-2 were significantly greater than concentrations at FMC. Sodium concentrations at VAL-1 and VAL-2 were significantly greater than concentrations at FMC. Concentrations of sodium, potassium, chloride, and sulfate at VIL-2 and VIL-3 were significantly greater than concentrations at VIL-1; concentrations of potassium at VAL-3 were significantly greater than at VAL-2.

Field and Continuous Measurements of Water Properties

Field measurements of physical properties of water, such as pH, temperature, specific conductance, and dissolved oxygen, can be used to compare chemical conditions in the streams at the time of sampling. Water quality, however, continually changes over time, and repeated measurements are necessary to characterize variations in quality. Continuous water-quality monitors have sensors and recording systems that measure water-quality properties at discrete time intervals and provide a continuous record of these properties over time.

The ADEM established criteria for pH, water temperature, dissolved oxygen, and turbidity based on water-use classification. Water-quality properties, including field measurements and continuous water-quality monitoring data, are summarized with applicable criteria in table 7. More detailed information obtained from continuous water-quality monitors in Village and Valley Creeks is summarized in the most recent USGS Annual Data Reports for Alabama (Pearman and others, 2001, 2002).

Long-term continuous water-quality data are available for two sites on Village Creek—VIL-3 since 1991 and VIL-4 since 1996 (table 1). In conjunction with this study, additional monitors were installed upstream from VIL-1 and at VAL-3 and turbidity probes were added to the monitors at VIL-1, VIL-3, and VAL-3. Continuous measurements of all four water-quality properties at VIL-1 and VIL-3 were recorded between April 2000 and February 2001; continuous measurements of all four properties at VAL-3 were recorded between August 2000 and February 2001 (table 7; Pearman and others, 2001, 2002).

The pH of surface water generally ranges from 6 to 9. When the pH falls below 4 or 5, possibly as a result of commercial or industrial discharges, urban runoff, acid mine drainage, or acid rain, the structure of the aquatic community may be affected. The ADEM established a pH range of 6 to 8.5 to reduce the effects of highly acidic or highly basic water on fish and wildlife (Alabama Department of Environmental Management, 2000d). Field measurements of pH were made at all sites at the time of sampling (appendix table 2-2). Continuous monitors, however, were not equipped to measure pH. In Valley Creek, pH ranged from 7.3 at VAL-1 to 8.5 at VAL-2; in Village Creek pH ranged from 6.9 at VIL-1 to 8.5 at VIL-3 (table 7). The pH values at all sites sampled were within the criteria established for agricultural and industrial water supply as well as fish and wildlife (Alabama Department of Environmental Management, 2000d). The higher pH measurements found at VIL-3, VAL-2, and VAL-3 are indicative of the carbonate-based geology in the area. The lowest pH measurements in both streams were recorded during storm events.

Specific conductance (SC) is an indicator of the ability of water to conduct an electric current and is proportional to the dissolved-solids concentration in water. Many factors affect the SC of streams, including flow conditions, bedrock geology, and contributions of dissolved solids from point and nonpoint sources. Standards or criteria for SC have not been established by the ADEM or the USEPA. SC was measured at the time of sample collection and by continuous water-quality monitors at four sites (table 7). In Valley Creek, field measurements of SC ranged from 57.5 to 599 microsiemens per centimeter at 25 degrees Celsius (μS/cm) at VAL-1. Continuous SC measurements ranged from 68 to 621 μS/cm at VAL-3. In Village Creek, field measurements of SC ranged from 76.3 μS/cm at VIL-1 to

Table 6. Results of the Kruskal-Wallis test and the Tukey multiple-comparison test illustrating statistically significant ($p \leq 0.05$) differences for selected water-quality constituents at sites in the Birmingham area, Alabama

[BOD, biochemical oxygen demand; °C, degree Celsius; *E. coli, Escherichia coli*]

Site label (fig. 1)	Water temperature	Discharge	pH	Specific conductance	Dissolved oxygen	Alkalinity	Total organic carbon	BOD	Hardness	Calcium
VIL1 to VIL2		X				X	X			
VIL2 to VIL3					X					
VIL1 to VIL3		X				X				
VAL1 to VAL2										
VAL2 to VAL3										
VAL1 to VAL3		X			X					
VIL1 to FMC										
VIL2 to FMC										
VIL3 to FMC										
VAL1 to FMC										
VAL2 to FMC				X						
VAL3 to FMC										

Site label (fig. 1)	Magnesium	Sodium	Potassium	Chloride	Sulfate	Fluoride	Silica	Solids, residue at 180 °C	Solids, sum of constituents, dissolved	Total nitrogen
VIL1 to VIL2		X	X	X	X	X				
VIL2 to VIL3										
VIL1 to VIL3	X	X	X	X	X	X				
VAL1 to VAL2										
VAL2 to VAL3			X							
VAL1 to VAL3						X				X
VIL1 to FMC	X									
VIL2 to FMC			X	X	X	X				
VIL3 to FMC			X	X	X	X				X
VAL1 to FMC		X	X	X	X	X				X
VAL2 to FMC		X	X	X	X	X		X	X	X
VAL3 to FMC			X		X	X				

Site label (fig. 1)	Dissolved nitrogen organic	Total nitrogen organic + ammonia	Dissolved nitrogen organic + ammonita	Dissolved nitrogen ammonia	Dissolved nitrite	Dissolved nitrate	Dissolved nitrite + nitrate	Total phosphorus	Dissolved phosphorus	Dissolved ortho phosphate
VIL1 to VIL2	X	X	X		X			X		
VIL2 to VIL3					X					
VIL1 to VIL3	X		X	X	X					
VAL1 to VAL2		X	X	X	X			X	X	X
VAL2 to VAL3										
VAL1 to VAL3	X	X	X	X	X			X	X	X
VIL1 to FMC						X	X			
VIL2 to FMC			X	X	X			X	X	X
VIL3 to FMC			X	X	X					
VAL1 to FMC	X	X	X	X	X	X	X	X	X	X
VAL2 to FMC				X	X			X	X	X
VAL3 to FMC									X	X

$p > 0.05$ No statistically significant differences between sites as determined by the Kruskal-Wallis test.

X $p \leq 0.05$ Statistically significant differences between sites as determined by the Tukey multiple-comparison test (nonparametric).

$p > 0.05$ No statistically significant differences between sites as determined by the Tukey multiple-comparison test (nonparametric).

Table 6. Results of the Kruskal-Wallis test and the Tukey multiple-comparison test illustrating statistically significant ($p \leq 0.05$) differences for selected water-quality constituents at sites in the Birmingham area, Alabama—Continued

[BOD, biochemical oxygen demand; °C, degree Celsius; *E. coli, Escherichia coli*]

Site label (fig. 1)	Suspended phosphorus	Dissolved nonortho-phosphorus	Nitrogen load	Nitrogen yield	Phosphorus load	Phosphorus yield	Fecal coliform	*E. coli*	Enterococci	Aluminum
VIL1 to VIL2	X	X			X					
VIL2 to VIL3										
VIL1 to VIL3			X		X					
VAL1 to VAL2	X	X					X	X		
VAL2 to VAL3										
VAL1 to VAL3	X	X					X	X		
VIL1 to FMC										
VIL2 to FMC										
VIL3 to FMC										
VAL1 to FMC	X	X					X	X		
VAL2 to FMC		X								
VAL3 to FMC										

Site label (fig. 1)	Barium	Cadmium	Copper	Iron	Lead	Manganese	Molybde-num	Zinc	Simazine	Prometon
VIL1 to VIL2	X	X					X			
VIL2 to VIL3										
VIL1 to VIL3	X						X			
VAL1 to VAL2	X									
VAL2 to VAL3										
VAL1 to VAL3										
VIL1 to FMC										
VIL2 to FMC	X	X					X			
VIL3 to FMC	X		X				X			
VAL1 to FMC			X							
VAL2 to FMC							X			X
VAL3 to FMC										

Site label (fig. 1)	Diazinon	Atrazine	Number of pesticide detections	Wastewater indicators						
				Food by-products	Pharma-ceuticals	Phosphates	Detergents	Fragrance	Total concen-tration	Number of detections
VIL1 to VIL2						X				X
VIL2 to VIL3										
VIL1 to VIL3						X		X		X
VAL1 to VAL2				X						X
VAL2 to VAL3										
VAL1 to VAL3				X	X	X		X		X
VIL1 to FMC										
VIL2 to FMC						X	X		X	X
VIL3 to FMC						X	X		X	X
VAL1 to FMC				X	X	X	X		X	X
VAL2 to FMC						X	X		X	X
VAL3 to FMC										

$p > 0.05$ No statistically significant differences between sites as determined by the Kruskal-Wallis test.

X $p \leq 0.05$ Statistically significant differences between sites as determined by the Tukey multiple-comparison test (nonparametric).

$p > 0.05$ No statistically significant differences between sites as determined by the Tukey multiple-comparison test (nonparametric).

Table 7. Water-quality properties of streams in the Birmingham area, Alabama, 2000–01

[Values shown in **bold** exceeded the criteria. µS/cm, microsiemens per centimeter at 25 degrees Celsius; °C, degree Celsius; mg/L, milligrams per liter; NTU, nephelometric turbidity unit; —, no criteria established; ≤, equal to or less than; ≥, equal to or greater than; ND, no data were collected]

Site label (fig. 1)	pH	Specific conductivity (µS/cm)	Water temperature (°C)	Dissolved oxygen (mg/L)	Specific conductivity (µS/cm)	Water temperature (°C)	Dissolved oxygen (mg/L)	Turbidity (NTU)
Water-quality criteria for agricultural and industrial water supply[a]								
	6.0–8.5	—	≤ 32.2	≥ 3.0	—	≤ 32.2	≥ 3.0	≤ 50[b]
Water-quality criteria for fish and wildlife[a]								
	6.0–8.5	—	≤ 32.2	≥ 5.0[c]	—	≤ 32.2	≥ 5.0[c]	≤ 50[b]
	Field measurements				**Continuous water-quality data**			
VIL-1	6.9–8.1	76.3–393	11.3–24.0	7.0–9.2	54–403[d]	7.4–29.7[d]	**2.8**–15.8[d]	0.5–870[d]
VIL-2	7.0–8.1	114–760	10.3–26.2	5.4–10.5	ND	ND	ND	ND
VIL-3	7.2–8.5	184–467	9.9–29.2	5.9–11.8	77–614	1.7–**33.5**	**1.7**–17.7	1.0–1,900[e]
VIL-4	7.2–7.4	144–510	15–25.8	**0.0**–8.6	81–587	8.1–30.7	**2.7**–12.7	ND
VAL-1	7.3–8.0	57.5–599	10.9–25.1	**3.3**–10.4	ND	ND	ND	ND
VAL-2	7.5–8.5	126–539	5.5–30.0	**4.3**–13.1	ND	ND	ND	ND
VAL-3	7.6–8.4	103–489	7.0–27.9	6.5–13.9	68–621	1.9–31.3	**3.8**–19.6	0.1–100
LCR	7.5–8.0	296–403	14.6–23.3	6.6–9.9	ND	ND	ND	ND
FMC	7.4–8.2	128–364	8.5–24.4	6.6–12.2	ND	ND	ND	ND

[a] Criteria established by the Alabama Department of Environmental Management (2000d).
[b] Turbidity will not exceed 50 NTU above background (Alabama Department of Environmental Management, 2000d).
[c] Criteria for dissolved oxygen under extreme conditions is 4.0–5.0 mg/L (Alabama Department of Environmental Management, 2000d).
[d] Continuous water-quality monitor located upstream from site VIL-1 at station 02458148.
[e] Instrument range from 0 to 1,000 NTU.

760 µS/cm at VIL-2. Continuous SC measurements ranged from 54 µS/cm at VIL-1 to 614 µS/cm at VIL-3. The lowest specific conductance measurements in both streams were made during storm events and are a result of dilution by rainwater.

Dissolved-oxygen (DO) concentration is widely used for evaluating the biochemistry of streams and lakes. DO concentrations may be depleted by processes that consume organic matter. Actively photosynthesizing algae and aquatic plants can increase concentrations of DO (Hem, 1985). The ADEM established criteria for DO concentrations in streams based on water-use classification (Alabama Department of Environmental Management, 2000d). For diversified warm-water biota, daily DO should not fall below 5 mg/L. Under extreme conditions resulting from natural causes, DO may range from 4.0 to 5.0 mg/L provided that the water quality is favorable in all other properties (Alabama Department of Environmental Management, 2000d). In streams classified for agricultural and industrial use, daily DO should not fall below 3.0 mg/L (Alabama Department of Environmental Management, 2000d). Low concentrations are commonly found in waters that are warm and not well mixed. DO concentrations typically vary in a diurnal fashion, and differences between high and low values can exceed 10 mg/L within a 24-hour period (Pearman and others, 2001). An example of the diurnal fluctuation of DO at VIL-3 is illustrated in figure 9. During low-flow conditions (August 27–31, 2000) peak DO concentrations (11.9–13.3 mg/L) were recorded between 4 and 6 p.m. (1600 and 1800 hours, fig. 9) each day, and minimum DO concentrations (3.6–5.4 mg/L) were recorded between 5 and 6 a.m. (0500 and 0600 hours, fig. 9) each day.

During this study, DO did not always remain within levels established by the ADEM. In Valley Creek, DO remained above the criterion established for agricultural and industrial water supply, but was less than the criterion established for fish and wildlife (table 7). In Valley Creek, field measurements of DO ranged from 3.3 mg/L at VAL-1 to 13.9 mg/L at VAL-3. Continuous measurements of DO ranged from 3.8 to 19.6 mg/L at VAL-3. Concentrations between 4.0 and 5.0 mg/L were measured in the field twice at VAL-1 and twice at VAL-2. Daily minimum concentrations between 4.0 and 5.0 mg/L were recorded by the continuous water-quality monitor at

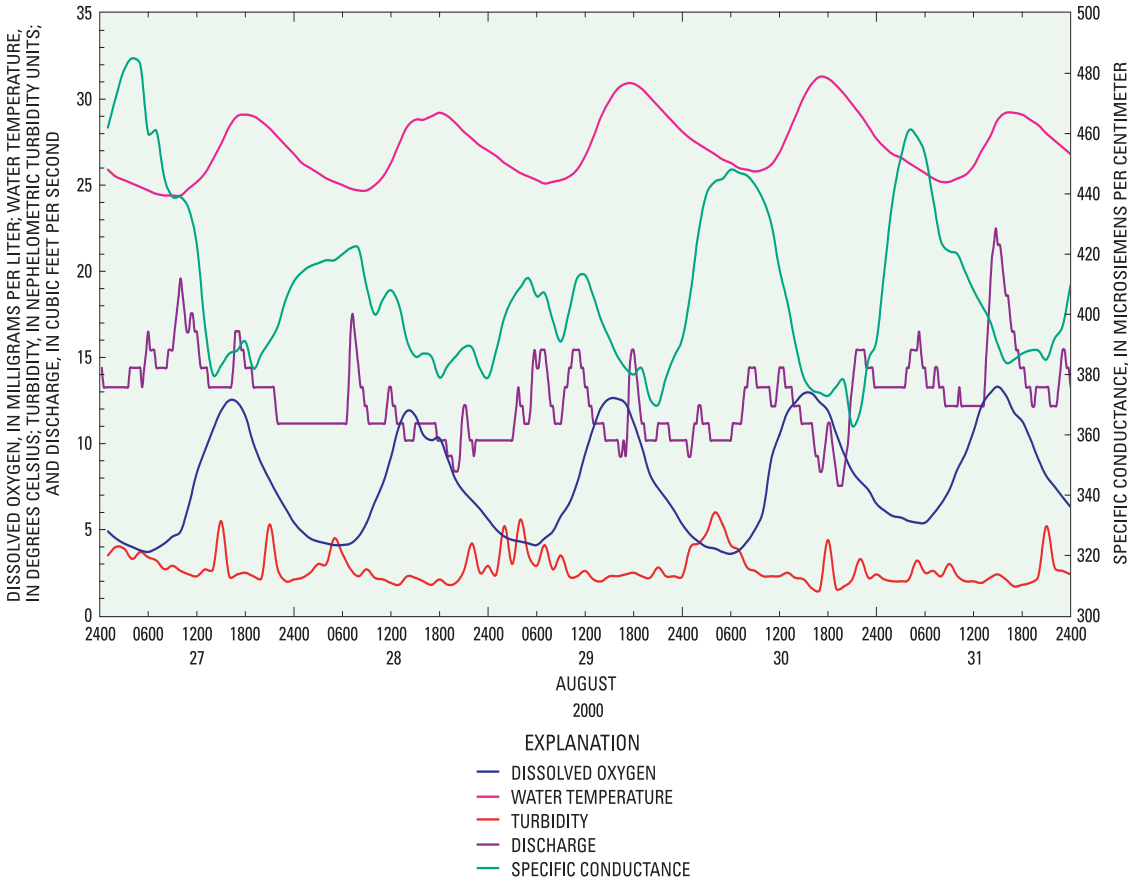

Figure 9. Dissolved oxygen, water temperature, turbidity, discharge, and specific conductivity at U.S. Geological Survey streamgaging station 02458450 (VIL-3) during low-flow conditions, August 27–31, 2000.

VAL-3 on 39 separate days between June 15, 2000, and February 22, 2001; concentrations less than 4.0 mg/L were recorded on 1 day during this time period.

In Village Creek, DO levels were recorded that were less than the minimum criteria established for both agricultural and industrial water supply and for fish and wildlife (table 7). Field measurements of DO ranged from 0.0 mg/L at VIL-4 to 11.8 mg/L at VIL-3. Continuous DO measurements ranged from 1.7 to 17.7 mg/L at VIL-3. At VIL-1, daily minimum DO concentrations ranged between 4.0 and 5.0 mg/L on 2 separate days between April 26, 2000, and February 21, 2001; concentrations were less than 4.0 mg/L on 8 days and less than 3.0 mg/L on 4 days. At VIL-3, daily minimum DO concentrations ranged between 4.0 and 5.0 mg/L on 53 separate days between April 1, 2000, and March 31, 2001; concentrations were less than 4.0 mg/L on 35 days and less than 3.0 mg/L on 7 days. At VIL-4, daily minimum DO concentrations ranged between 4.0 and 5.0 mg/L on 53 separate days between April 1, 2000, and March 31, 2001; concentrations were less than 4.0 mg/L on 17 days and less than 3.0 mg/L on 2 days. On occasion, continuous DO data collection was interrupted because of technical difficulties in the field.

Turbidity, a measure of water clarity, is determined by measuring the degree that particles suspended in water decrease the passage of light through the water. Particles may come from soil, sediment, algae, plankton, natural organic matter, or manmade compounds. High turbidities are commonly measured during storms when overland runoff erodes soil and carries it to the stream, and increased flow resuspends sediment in the streambed. However, high turbidities can also be measured during low flow when certain materials or compounds are discharged from industrial and commercial facilities. The ADEM criterion requires that turbidity not exceed 50 nephelometric turbidity units (NTU) above background except due to natural origin (Alabama Department of Environmental Management, 2000d). Background levels have not been defined by the ADEM for Village and Valley Creeks; consequently, turbidity values were not compared to this criterion.

Continuous measurements of turbidity in Village and Valley Creeks varied with streamflow (Pearman and

others, 2001, 2002). At VAL-3, turbidity ranged from 0.1 to 100 NTU. At VIL-1, turbidity ranged from 0.5 to 870 NTU. At VIL-3, turbidity ranged from 1.0 to 1,900 NTU (table 7). High turbidities were observed at both streams and in many cases, likely were the result of natural runoff, but in some instances, particularly at Village Creek, these high turbidities may be attributed to anthropogenic causes. At Valley Creek, high turbidity was consistently measured during high flow. At Village Creek, high turbidity was measured during low flow as well as high flow, which may indicate the presence of point or other anthropogenic sources.

Turbidity also can be used to examine whether high-flow samples were collected during the first flush, when many contaminants may be at a maximum level. Continuous water-quality data were recorded over a 4-day period at VIL-3 (August 1–4, 2000) during which two storms occurred (fig. 10). During the first storm (August 2, 2000), turbidity peaked (915.2 NTU) at 3 p.m. (1500 hours, fig. 10)—while discharge peaked (1,829 ft^3/s) at 4:30 p.m. (1630 hours, fig. 10). The first flush most likely occurred around 3 p.m. (1500 hours, fig. 10), when turbidity values were highest. During the second storm (August 4, 2000), turbidity peaked (681.2 NTU) at 5 p.m. (1700 hours, fig. 10), and discharge peaked (1,003 ft^3/s) at 4:45 p.m. (1645 hours, fig. 10), illustrating the likelihood that there was no first flush associated with this storm, perhaps due to the preceding storm 2 days earlier. Turbidity values were recorded on an hourly basis at VIL-1, VIL-3, and VAL-3. Discharge was recorded every 15 minutes at VIL-1 and VIL-3.

Nutrients

In natural waters, nitrogen is a combination of different chemical forms, depending on the source and environmental conditions. Common forms include organic nitrogen, which can be in dissolved or particulate form, and the inorganic ions, ammonium (NH_4^+), nitrite (NO_2^-), and nitrate (NO_3^-), which are typically in dissolved form. The nitrogen cycle is a series of

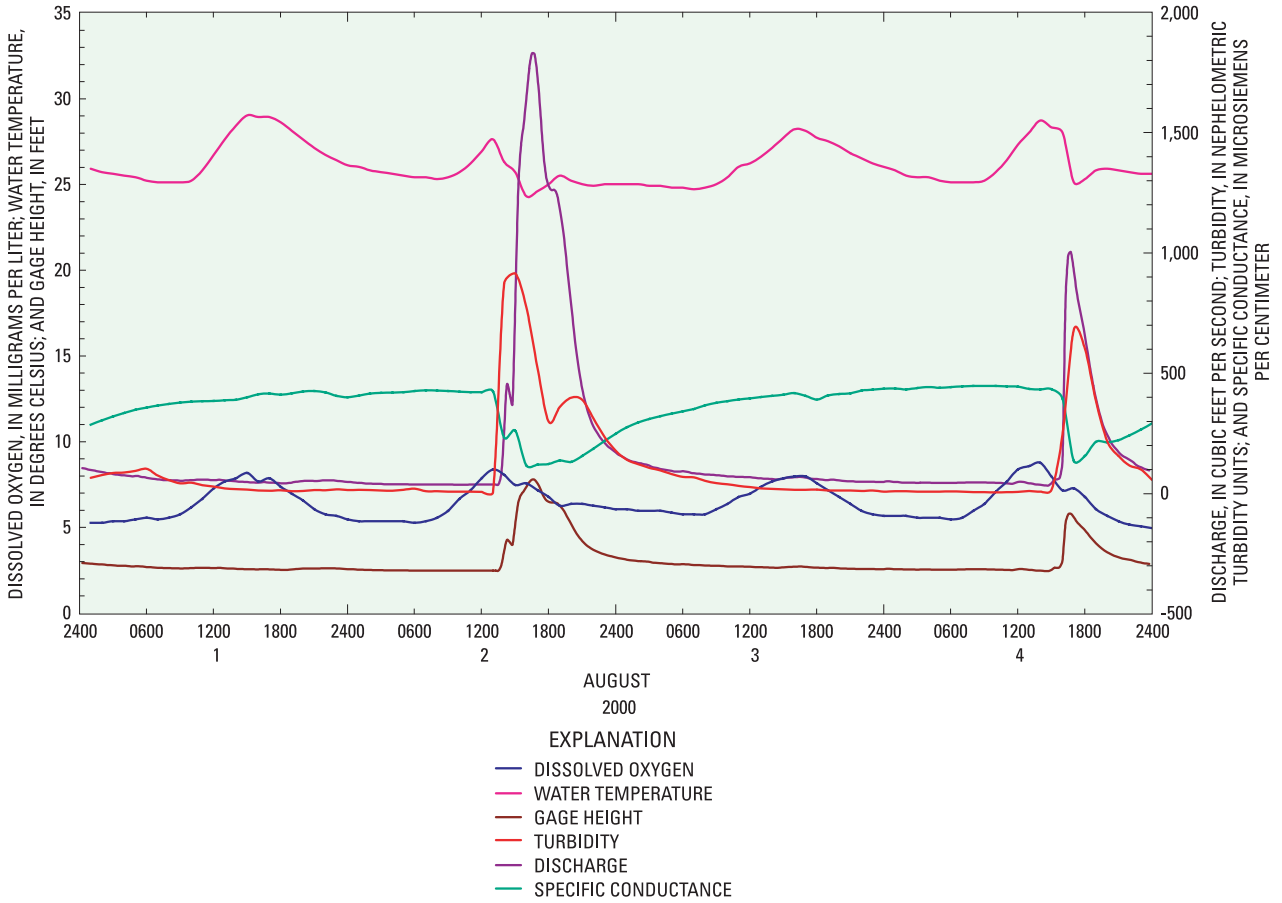

Figure 10. Dissolved oxygen, water temperature, gage height, discharge, turbidity, and specific conductivity at U.S. Geological Survey streamgaging station 02458450 (VIL-3) during two storms, August 1–4, 2000.