# Water Quality Concepts, Sampling, and Analyses

Edited by

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# 5 Surface Water Quality Sampling in Streams and Canals

*Kati W. Migliaccio, Daren Harmel, and Peter C. Smiley, Jr.*

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## 5.1 INTRODUCTION

Surface water sampling and water quality assessments have greatly evolved in the United States since the establishment of the Clean Water Act in the 1970s. Traditionally, water quality referred to only the chemical characteristics of the water and its toxicological properties related to drinking water or aquatic life uses, but now water quality includes physical, chemical, and biological characteristics. Surface water sampling reflects the changing views of water quality and the emerging variety of sampling goals. Surface water sampling projects range from very simplistic to highly complex and differ based on spatial extent, response variables analyzed, flow conditions sampled, and technology used. This chapter provides up-to-date information on designing and implementing surface water quality sampling projects in wadeable streams and in wadeable canals constructed to improve drainage characteristics. This chapter not only focuses on procedures for determining chemical quality but also provides introductory concepts related to evaluation of the physical and biological aspects of water quality.

## 5.2 DEFINING PROJECT GOAL

The first and often the most difficult step in designing a surface water sampling project is clearly identifying the project goal. This step must be accomplished because the goal ultimately determines sampling methods, sampling frequency, and response variables. Typical sampling goals include but are not limited to the following:

1. Meeting total maximum daily load (TMDL) monitoring requirements
2. Quantifying the performance of Best Management Practices (BMPs)
3. Identifying source(s) of constituent(s) of concern
4. Quantifying reference or background constituent concentrations
5. Determining if the quality of water in a water body is meeting designated uses

These few examples highlight that water sampling goals can range from simple descriptive assessments intended to document existing conditions to complex studies intended to determine watershed level impacts of specific practices. Thus, it is critical to recognize that different goals result in different sampling intensities and sampling designs (Smiley et al., 2009).

The primary limitation in design and implementation of any sampling project is resource availability or constraints. As such, the success of a surface water sampling project meeting its established goal is typically determined by its ability to accurately characterize water quality conditions with available resources (Harmel et al., 2006a). Others have reported the difficulty and importance of achieving this balance (e.g., Preston et al., 1992; Shih et al., 1994; Tate et al., 1999; Agouridis and Edwards, 2003; Harmel et al., 2003; King et al., 2005; Harmel and King, 2005). As with most multifaceted objectives, optimization is required to ensure that resources are effectively and efficiently allocated (Abtew and Powell, 2004; Miller, 2005). Factors that affect resource allocation should be carefully considered in project design and implementation and include factors such as site selection, equipment maintenance, personnel requirements, discharge measurement, water chemistry sampling methodology, and physical and biological assessment methodologies.

## 5.3 SITE SELECTION

Selection of the initial set of potential sampling sites is primarily based on the goal of the sampling project. Data collected from sampling sites should answer a question or hypothesis proposed by the sampling goal. For example, a sampling goal may be to quantify the reference or background concentration of a constituent in a watershed. The question that needs to be answered would be "What is the concentration of a particular constituent in the most downstream location (or outlet) of selected subbasins in the watershed that are minimally or not impacted by human influences?" Constituent data collected from sampling sites located at the most downstream location of the minimally impacted subbasins that represented reference conditions would be able to answer this question. Another sampling goal might be to determine if water quality in a water body was meeting designated uses. The question that needs to be answered would be: "How do water quality measurements collected from a stream compare to water quality criteria associated with the designated uses of that stream?" Data from sampling sites located strategically in the stream would be needed so that sampling sites were placed where concentrations were thought to be the greatest (and variable) due to point source inputs, tributary influences, or other known features of the system. Data collected would be compared to water quality criteria to determine if designated use standards were being met. Thus, site selection requires identification of goals and some understanding of the system processes, so appropriate sampling sites are selected and the collected data can answer the hypothesis or water quality question being posed.

Site selection also depends on the scale associated with the sampling goal. The greater heterogeneity inherent with watershed scale studies compared to small field-scale studies requires consideration of multiple point and nonpoint constituent sources and how these sources may impact the appropriateness of particular sites considering the sampling goal. This is especially important when evaluating water quality near effluent discharges from Waste Water Treatment Plants (WWTPs; Migliaccio et al., 2007; Carey and Migliaccio, 2009) and immediately downstream of tributary confluences. Site selection should also consider potential stream modification that may occur naturally or due to anthropogenic activity. Modifications often include widening or shifting of the channel and erosion of embankment. Hence, sites should be selected in areas that are anticipated to remain stable and where signs of impending modification are not present, but this may not always be possible as stream systems naturally meander over time. For these sites, the physical dynamics of the stream should be considered for site selection. Sites should also be located (if possible) at existing flow gauges or hydraulic control structures with an available historical flow record and established stage–discharge relationship because of the difficulty of establishing accurate stage–discharge relationships (discussed in Section 5.6).

Potential sites may be identified using aerial photographs, detailed maps, or personal knowledge of the area (Benson and Dalrymple, 1984). Technology and Internet tools have progressed, and now this task may also be completed using GIS software or even ready-to-use tools such as Google Earth. Once potential sites have been selected, field visits are needed to evaluate site characteristics to ensure that each site will support project goals and optimize available resources. The accessibility and integrity of each potential sampling site should also be carefully considered. Site characteristics such as personnel and vehicle access (especially in wet conditions), flood likelihood, adjacent land ownership, and vandalism potential must be considered in site selection. Sampling projects with long-term goals should also consider the probability of significant construction or land modification near the sampling site, especially if these activities significantly alter flow and constituent transport conditions. If significant impact is expected from such activities, alternative sites should be selected. Sites should also be selected to minimize travel expenses (USDA, 1996). Travel costs to and from sampling sites can be substantial, especially as distances increase, requiring more personnel time and increased transportation expense. Trips to distant sampling locations can be especially difficult and costly in wet periods when frequent trips are required to collect samples from automated sampling systems (Harmel et al., 2006a). Similarly, sample preservation and related quality assurance measures associated with sample transport become more difficult as time between retrieval and analysis increases. First-hand visitation of sites and assessment of these challenges will likely result in a reduction in the amount of feasible sites from the original list of potential sites.

Once the physical location of a site has been selected, the spatial extent at which the evaluation will be conducted at each site should be determined. The spatial extent of a site will differ depending on what types of variables are being measured. Water chemistry measurements and discharge measurements are collected from one spot or at multiple spots along one transect, and thus the site is traditionally a point. In contrast, a site for the collection of physical habitat variables and biological sampling consists of multiple sampling locations within a defined reach of a specific length. The differences in what constitutes a site relates to the differences in the sampling protocols for each type of response variable. Physical habitat variables and the biota differ spatially within even short distances and, to ensure adequate characterization of these variables, sampling must be conducted in longer length sites than that typically used for water chemistry or discharge measurements. Water chemistry and discharge at one location are assumed to be representative of the combined flux (discharge and constituent) from upstream contributing areas.

The considerations that must be included in defining a site for sampling physical habitats or biota variables are further examined by considering the following example related to fish sampling. A common method for determining site lengths for fish sampling involves measuring the wetted width of the stream and then multiplying the mean base flow wet width by a constant to determine the site length to sample (Lyons, 1992). Recommended constants range from 14 times (Patton et al., 2000) to 120 times (Paller, 1995) the wetted stream widths, and they differ among response variables of interest, ecoregions, and sampling protocols. They even differ among national assessments of water quality. The USGS National Water Quality Assessment uses 20 times wetted width with a minimum of 150 m long and a maximum of 300 m long in wadeable streams (Lazorchak et al., 1998). In contrast, the USEPA Environmental Monitoring and Assessment program uses 40 times wetted width with a minimum of 150 m (Moulton et al., 2002). This proportional distance method of establishing site lengths is intended to ensure the accurate determination of species composition within a site without oversampling or undersampling. However, it results in variable site lengths among different-sized streams where small streams have shorter sites than large streams. Thus, if water quality comparisons among streams, sampling periods, or different categories of streams are important, then fixed length sites would be preferable to standardize sampling efforts and ensure comparability (Smiley et al., 2009).

## 5.4 EQUIPMENT MAINTENANCE

It is inevitable that equipment malfunction will occur, which makes proper installation and operation extremely important to limit errors and missing data. Sampling equipment should be maintained per the manufacturer's instructions and usually includes calibration, cleaning, and general maintenance. Failure to adequately maintain automated samplers is by far the greatest cause for malfunction and missed data. The important maintenance requirements for automated samplers include battery level (under load), solar panel output, sample lines (clogs or holes), stage/flow recorder accuracy, and desiccant capacity (Harmel et al., 2006a). All of these should be checked weekly or every other week at a minimum. A maintenance log should be kept and include a schedule for maintenance checks throughout the life of the project to ensure that maintenance is not overlooked. Each maintenance check and calibration procedure should be recorded in the maintenance log. This process should be included as part of the quality assurance/quality control plan (see Chapters 4 and 14).

Data collected and stored on-site with loggers should regularly be downloaded to minimize data loss due to power supply problems, lightning, vandalism, and other mishaps. Real-time data transfer and communication with remote sites can minimize data loss and unnecessary site visits by notifying personnel when problems occur and when samples have been collected. We emphasize that real-time equipment does not eliminate the need for frequent site visits to conduct maintenance and collect samples. Backup equipment should be purchased in anticipation of equipment malfunction or failure to allow for quick replacement and minimal data loss.

Surface water sampling equipment, such as automatic samplers, stage and/or velocity devices, data loggers, and power supplies, should be installed with consideration to environmental factors (i.e., plants, animals, vandals, and weather) and protected from damage that would alter sample integrity. All equipment should be housed in locked, sturdy structures positioned above the highest expected flow elevation to ensure accessibility during high flows (Haan et al., 1994; USEPA, 1997).

## 5.5 PERSONNEL REQUIREMENTS

The most critical component in a surface water sampling is the personnel. Personnel should be trained during project initiation to perform the necessary tasks and should receive refresher training throughout the project. It is best to have two-person teams dedicated to particular sampling aspects to minimize safety concerns and to ensure continuous staffing. It is best not to have multiple people collecting data separate from each other to limit personnel biases and/or technique inconsistencies.

Water sampling requires dedicated personnel and long hours of work often outside of conventional working hours; thus, project design should include estimation of personnel hours. Different sampling strategies (such as manual sampling versus automatic sampling) require different personnel needs and result in different costs. In addition, personnel will often need to travel to sites during adverse weather conditions; therefore, the time needed to complete tasks may be longer than if conditions were optimum.

## 5.6 DISCHARGE MEASUREMENT

The primary hydrologic component measured in surface water sampling projects is discharge. Discharge refers to the flow rate of water as measured in units of volume per time, such as gallons per minute (gpm) or cubic meters per second ($m^3\ s^{-1}$). Collection of appropriate discharge data is essential to adequately characterize water quality in terms of (1) offsite constituent loss, (2) downstream constituent transport, and (3) channel erosion and deposition. In addition, discharge data and associated constituent concentrations are needed to determine constituent mass flux (or loads) and to differentiate among transport mechanisms (Harmel et al., 2006a).

### 5.6.1 Continuous Discharge Measurement

Continuous discharge values are measured using monitoring instrumentation that is installed in the field and automatically recorded by a logging device. Such sites, often referred to as *gauging stations*, are often ideal water quality sampling locations. This is because colocation of sites allows for load calculations (from the product of water quality concentrations and discharge). Discharge measurements are typically stored in the field and collected using remote or on-site data transfer methods. The most common continuous discharge measurement method utilizes the stage–discharge relationship (also known as a rating curve). The basic premise of this method is that stage (water surface level or water depth) is continuously recorded and used to estimate the discharge based on an established mathematical relationship depicted by the rating curve. Frequent adjustments to this relationship are necessary in unstable channels to minimize the uncertainty in discharge data.

An established stage–discharge relationship accompanies precalibrated hydrologic structures such as flumes or weirs. For small watershed sites, precalibrated flow control structures are useful because they provide reliable and accurate flow data (Slade, 2004). However, weirs and flumes are expensive to purchase and install, and may result in flow ponding that impacts sediment transport (USDA, 1996). Another disadvantage is that precalibrated structures are limited in the discharge they support, which restricts their use as watershed size increases. Improper installation resulting in incomplete flow capture, unlevel alignment, or inappropriate approach channel characteristics can introduce considerable uncertainty (Rantz, 1982).

For sites where precalibrated structures are not feasible, a stage–discharge relationship can be established from a series of instantaneous stage and discharge measurements (discussed subsequently). Since stage and discharge data must be collected for the range of expected values, developing a stage–discharge relationship is a time consuming, long-term task.

Once a stage–discharge relationship has been established, sensors are used to continuously measure the stage. Common sensor types include bubblers, pressure transducers, noncontact sensors, and floats (Buchanan and Somers, 1982; USDA, 1996). Bubblers and pressure transducers are submerged sensors that estimate water stage by sensing the pressure head created by water depth. Noncontact sensors are suspended above the water surface and use ultrasonic or radar technology to measure surface water level (Costa et al., 2006). Float sensors float on the water surface and, in conjunction with a stage recorder, produce a graphical or electronic data record of stage. To increase the accuracy of stage measurement, stage sensors should be installed in a stilling well for protection and creation of a uniform water surface. Sensors must also be calibrated using a surveyed point for establishing correct stage measurements (Brakensiek et al., 1979; Haan et al., 1994). Installation of a permanent staff gauge is also recommended (USDA, 1996).

Although stage–discharge relationships are most often used for continuous discharge measurement, instream velocity meters and stage sensors are also appropriate alternatives. These instruments use multiple velocity measurements and corresponding stage data with cross-sectional survey data to determine the cross-sectional flow area and discharge (Harmel et al., 2006a). These devices may be preferred for sites where rapid morphological change or bidirectional flow occurs or where a structure is not feasible (e.g., waterways).

Continuous discharge may also be determined using Manning's equation (Maidment, 1993; Haan et al., 1994). Manning's equation estimates flow velocity using physical features of the system: channel roughness, slope, and cross-sectional geometry. Cross-sectional survey data are used with the velocity estimate to determine discharge. This method, however, introduces substantial uncertainty into discharge data as it was developed for uniform flow and because accurate channel roughness coefficients are difficult to select (Maidment, 1993). Thus, Manning's equation should only be used as a last alternative when other methods are not feasible for estimation of continuous discharge data.

### 5.6.2 Noncontinuous (Instantaneous) Discharge Measurement

Noncontinuous discharge measurements are collected using portable equipment that is not permanently installed in the field. Instantaneous discharge is often determined with the area–velocity method using velocity measurements and cross-sectional flow areas. The area–velocity method requires that water depth and velocity be measured in multiple vertical sections (each with no more than 5% to 10% of total flow) perpendicular to water flow in order to calculate mean velocity and cross-sectional area for each section (Buchanan and Somers, 1976; Figure 5.1). Velocity within the vertical profile of the stream is not uniform; thus, it is recommended that velocities be collected from 0.2 and 0.8 depths and averaged if the depth is greater than 0.6 m. Otherwise, velocity measurements are taken at the 0.6 depth to determine the mean velocity for the area–velocity method (Chow et al., 1988). The total discharge ($Q$) is the sum of discharges for each section and is calculated as

$$Q = \sum_{i=1}^{n} \left( (X_{i+1} - X_i)(U_i Y_i + U_{i+1} Y_{i+1}) / 2 \right) \tag{5.1}$$

where $n$ is the number of sections, $X$ is the distance measurement on the horizontal scale, $U$ is the velocity, and $Y$ is the depth measured on a vertical scale.

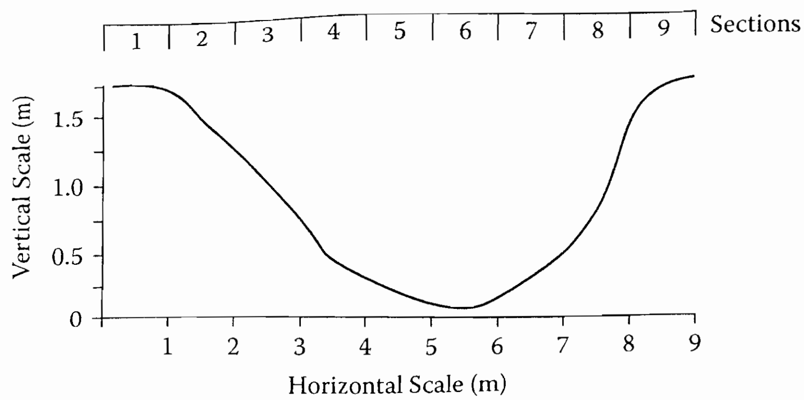


**FIGURE 5.1** The stream bottom (the line) is presented with vertical and horizontal scale axis; each vertical section is measured for depth, width, and velocity; and the resulting measurements are used to calculate flow (discharge) using the area–velocity method.

Several portable devices are available to measure water velocity. Velocity meters or current meters may use revolving cups that spin at a rate proportional to the velocity, or they may use Doppler, electromagnetic, or radar technology to determine flow velocity (Rantz, 1982; Morlock, 1996).

Another instantaneous discharge method involves dye or tracer studies that can be used where stream geometry prevents the application of the area–velocity method. Dye and tracer studies are useful for understanding advection and longitudinal dispersion, groundwater inflow, and hydrologic retention properties of a stream (Lee, 1995; Harvey and Wagner, 2000).

The drift method is another noncontinuous discharge measurement alternative that might be considered to calculate discharge if safety concerns or costs prevent the area–velocity method. This method consists of watching and timing a floating device as it travels a specified distance in stream flow. This method is the least accurate and should be considered the least preferable option. When using this method, the floating device should be selected so that it floats with the majority of its volume underwater to prevent wind influences on velocity measurement. Drift velocity measurements should be collected by positioning the floating device in the middle of the stream and reducing the velocity value determined by approximately 60%. The discharge can then be determined as the product of the cross-section and the velocity (Burton and Pitt, 2002).

## 5.7 WATER CHEMISTRY SAMPLING

Water chemistry sampling refers to the activity of collecting a sample of water from a selected water body that represents the constituent concentration of that water body with the intent of analyzing the sample for selected constituent concentrations. Water chemistry sampling may also refer to the collection of a water characteristic (such as temperature, conductivity, or dissolved oxygen) in situ using portable sensors. Sampling project design should consider key elements (processes) of surface water systems that might hinder collection of a representative sample and attempt to minimize potential biases. The following sections briefly describe key sample collection considerations.

### 5.7.1 Sample Collection Point

Although the location of sample collection can result in considerable differences in constituent concentrations (Martin et al., 1992; Ging, 1999), this issue is commonly ignored in many sampling projects, especially those using automated samplers with a single intake. It is generally assumed that dissolved constituents can be adequately sampled at a single location within the flow cross-section for field-scale sites and small streams unless immediately downstream of significant point sources (Martin et al., 1992; Ging, 1999; Slade, 2004). However, recent research indicates that dissolved constituents may exhibit horizontal and/or vertical variability within the cross-section (Harmel, unpublished data). The assumption of well-mixed conditions can be evaluated by measuring pH, temperature, conductivity, and dissolved oxygen throughout a stream cross-section. If collected measurements differ by less than 5% throughout the cross-section, then a single measurement point at the centroid of flow adequately represents the cross-section (Wilde and Radtke, 2005). The degree to which constituents are distributed within the flow cross-section is a major component in the uncertainty associated with sample collection at a single location (see Chapter 12 regarding measurement uncertainty).

In sites that exhibit considerable concentration variability within the flow cross-section, sampling methods should be adjusted to account for this variability. Multiple grab samples can be collected, but integrated techniques (discussed in Section 5.7.3) are the preferred manual sampling technique. For automated sampling, the recommended methodology involves development of a relationship between concentrations at the sampler intake and mean concentrations as determined by integrated sampling at a range of discharges (e.g., Ging, 1999). With such a relationship, concentrations at the intake can be adjusted to represent mean concentrations for the total cross-section.

In contrast to dissolved constituents, sediment and sediment-bound constituent concentrations almost always vary substantially within the flow cross-section. For field-scale and small watershed sampling locations, a single sample intake may be adequate for sediment sampling because of well-mixed conditions and shallow water depths. Sediment sampling of coarse sediment or sediment in larger streams requires integrated sampling to adequately capture sediment concentration variability (Harmel et al., 2006a). Water quality sampling for sediment and sediment-associated constituents is more difficult than sampling for other water chemistry variables because of temporal and spatial variability in transport. Sediment particles <62 μm in diameter are generally homogeneously distributed throughout a channel's cross-section (Vanoni, 1975; Edwards and Glysson, 1999). However, particles >62 μm typically exhibit vertical and horizontal concentration gradients with the greatest concentration near the stream bed, and concentrations decrease with increasing distance from the bed. Therefore, the point at which a sample is collected will influence sediment concentration data. Samples for sediment sizes >62 μm should be collected with integrated techniques or should be adjusted to represent mean cross-sectional concentrations. Sediment sizes between 62 to 2,000 μm should be sampled using isokinetic techniques. Isokinetic sampling refers to collection of a water-sediment mixture in such a way that there is no change in velocity as the sample leaves the ambient flow and enters the sampler intake. Equipment for taking isokinetic samplers has been developed and evaluated by the Federal Interagency Sedimentation Project (Edwards and Glysson, 1999). Other techniques are available for sampling sediment concentrations, such as acoustic backscatter and optical backscatter (Wren et al., 2000; Gray, 2005). While these alternatives collect more detailed data in a temporal sense and may produce more accurate data, standardized methods and applicability to various conditions are not well established (Gray and Glysson, 2003; Gray, 2005).

While the previous discussion focused on collection of water samples for analysis in the laboratory, other technologies are available for in situ estimation of water quality parameters. Technological advances have resulted in greater application of portable sensors for measuring water parameters in situ. These devices are commonly used for measurements of physicochemical variables (pH, water temperature, dissolved oxygen, conductivity), but advancements in biosensor development have resulted in a broader application of such portable sensors (Glasgow et al., 2004). Sensors range from simple pen-type instruments capable of on-the-spot measurements of pH lacking data storage capabilities to multiparameter meters designed for continuous measurements of a dozen or more water chemistry parameters including those typically measured in the laboratory (i.e., chlorophyll, blue-green algae, rhodamine, nitrate, ammonium, and chlorine). Many commercial vendors enable multiparameter meters to be custom fitted with selected probes to meet specific research needs. Chapter 8 discusses field measurement methods and equipment in greater detail.

### 5.7.2 Flow Regime Considerations

Water samples may be collected during base flow and/or storm flow conditions, depending on the goal of the sampling project, and it should be expected that substantial differences in concentrations will occur between these discharge regimes due to differing source contributions.

#### 5.7.2.1 Base Flow

Base flow is defined as the discharge derived from the seepage of groundwater in combination with upstream water through-flow without significant direct contribution from surface runoff resulting from precipitation. In small streams and canals, point source discharge can be an important base flow component.

Base flow water sampling is necessary at intermittent and perennial flow sites to quantify the contributions of point sources, tile drainage, shallow subsurface return flow, and constituent release from instream processes. Whereas base flow sampling is often needed in watershed-scale sampling projects, it can be unnecessary in field-scale projects that are usually conducted in runoff-dominated ephemeral sites (Harmel et al., 2006a). Since point sources may impact base flow concentrations, samples should be taken a distance from effluent discharges where flow is expected to be well mixed. Base flow samples collected at a single point in well-mixed flow, usually in the centroid of flow, are typically assumed to accurately represent true mean cross-sectional concentrations (Martin et al., 1992; Ging, 1999; Slade, 2004).

#### 5.7.2.2 Storm Event Sampling

Storm event sampling refers to sampling when discharge conditions are influenced by surface runoff due to a precipitation event. Such events are typically defined by a specific amount of precipitation occurring after a specific interval of no precipitation. Depending on the site location and sampling goal, the definition of a storm event will differ. The National Pollutant Discharge Elimination System (NPDES) Storm Water Sampling Guidance Document (USEPA, 1992) defines a storm event as the occurrence of 2.54 mm of accumulated precipitation after 72 h of preceding dry weather. The guidance also suggests that "where feasible, the depth of rain and duration of the event should not vary by more than 50% from the average depth and duration." It is important to note that this definition is specific to storm water sampling to meet permit requirements and that increased discharge containing surface runoff contributions may or may not occur following this amount of rainfall. Adjustments from this protocol may be implemented depending on project sampling objectives.

Another strategy for defining and identifying storm event flows is to evaluate discharge records (Institute of Hydrology, 1980a, 1980b). This method separates base flows and storm flows in a dataset using the principle that if 70% or greater of total discharge for a day is composed of base flow, then it is considered base flow. Otherwise, the discharge regime is characterized as a storm event flow. The limitation to this method is that it characterizes discharge measurements as base flow or storm flow after the data have been collected. Whatever method is selected, it is important to establish the storm definition at the beginning of the project to ensure that adequate samples are collected to characterize base flow and storm flow as required by the project goals.

Storm flow is often the focus of sampling projects because of its increased transport ability for recently washed off and resuspended constituents that have been attenuated by instream processes. Increased constituent transport in storm events can occur early in storms, and this phenomenon is often referred to as *first flush*. Alternatively, constituent concentrations can peak with peak flow, increase throughout storm events, or remain relatively uniform as storm and base flow contributions interplay. In any case, storm flows should be sampled throughout events to ensure proper water chemistry characterization (Huber, 1993). The unique circumstances of storm event sampling and the resulting strategies for implementing water quality sampling have only recently been described (e.g., McFarland and Hauck, 2001; Harmel et al., 2003; Haggard et al., 2003; King and Harmel, 2003; Behrens et al., 2004; Harmel et al., 2006a). Characterization of storm water chemistry is much more difficult than base flow because storm runoff often occurs with little advance warning, outside conventional work hours, and under adverse weather conditions (USEPA, 1997).

### 5.7.3 Sampling Equipment Considerations

Surface water samples from small streams and canals are collected manually or by an automated sampler permanently or semipermanently installed at the sampling location and programmed to collect water samples at desired intervals. Each method has its advantages and disadvantages.

#### 5.7.3.1 Manual Sampling

Manual grab sampling at a single location in the flow cross-section at each sampling site is the standard method for collection of base flow samples. Grab sampling has several advantages. It is relatively safe, simple, and inexpensive, can be performed at any location, and is not subject to equipment theft or vandalism. Unfortunately, grab sampling provides limited information on temporal variability of constituent concentrations unless frequent samples are collected. Manual sampling methods also introduce human errors due to sampling variability that might occur. Manual grab sampling may, however, be the only alternative if the capital costs of purchasing and maintaining automatic sampling equipment exceeds available resources (Burton and Pitt, 2002).

Integrated sampling is an alternative manual technique that collects subsamples throughout the flow cross-section to accurately determine mean cross-sectional constituent concentrations. Integrated sampling typically utilizes the USGS equal-width increment or equal-discharge increment procedures (Wells et al., 1990; USGS, 1999). With these procedures, multiple depth-integrated samples are obtained across the stream cross-section and have been shown to produce accurate concentration measurements even in large streams. However, these techniques require substantial personnel time, especially for multiple sites, and can be difficult for sample collection throughout the range of observed discharges. If human entry into the stream or canal is necessary for sample collection, personnel safety must be the utmost priority. If entry is safe, then streambed and bank disturbances should be limited to limit constituent resuspension. Similarly, water samples should be collected upstream from the point of entry.

#### 5.7.3.2 Automatic Sampling

Automated samplers offer several advantages to manual sampling. A major advantage of automated samplers is their ability to use consistent sampling procedure and simultaneously collect samples at multiple sites. Automatic samplers are advantageous for distant, hard-to-reach (e.g., steep inclines), and dangerous sites due to wildlife (e.g., alligators) or storm conditions (e.g., high flow, lightning). Automatic samplers are particularly useful for storm event sampling because of their ability to sample throughout runoff events of various durations and magnitudes. However, automated samplers also have some disadvantages such as their single sample intake and impossibility of keeping the intake in the centroid of flow (Harmel et al., 2006a). In addition, automatic samplers require frequent maintenance (Burton and Pitt, 2002; Harmel et al., 2006a).

Mechanical samplers, such as the rotating slot sampler and the multislot divisor sampler, may be practical alternatives to electronic automatic samplers, but they commonly have a discharge rate limitation at which they can effectively sample. Mechanical samplers collect flow-weighted samples and estimate flow volume, allowing for the calculation of event mean concentrations and mass loads. The rotating slot sampler requires minimal maintenance, no electrical power, and collects a single flow-proportional runoff sample (Parsons, 1954, 1955; Edwards et al., 1976). Others have modified this design for their specific application needs (Bonta, 1999; 2002; Malone et al., 2003). The multislot divisor has also been applied by many investigators for collecting surface runoff from fields or slopes (Geib, 1933; Sheridan et al., 1996; Franklin et al., 2001; Pinson et al., 2003).

## 5.8 PHYSICAL HABITAT AND BIOLOGICAL ASSESSMENTS

Stream ecosystems are complex, dynamic systems and often physical, biological, and chemical variables are interrelated. Thus, understanding these relationships can help with developing watershed management recommendations. Physical habitat and biological assessments are an integral part of water quality monitoring programs in the United States as all states have a monitoring program that includes physical habitat, biological, and traditional water chemistry and discharge assessments (USEPA, 2002). Many stream sampling protocols consist of both descriptions of physical habitat and biological sampling in the same protocol (USEPA, 2002; Somerville and Pruitt, 2004). These joint assessments are necessary as the physical and chemical characteristics of streams are needed to interpret the observed biological responses (see also Chapter 3, Section 3.4.2). More than 400 habitat and biological sampling protocols have been developed for a wide range of studies (Johnson et al., 2001; NRCS, 2001; USEPA, 2002; Somerville and Pruitt, 2004). However, few if any sampling protocols are universally accepted (Frissell et al., 2001). In this section, an introduction to key concepts and commonly used sampling methods for assessing physical habitat and biological characteristics of streams is provided.

### 5.8.1 Physical Habitat Assessments

Physical variables, such as channel cross-section area, discharge, number of riparian trees, and percent sand substrate, are often referred to by ecologists as physical habitat variables because they serve as descriptors of the space or the "habitat" that fishes, insects, aquatic plants, and other stream organisms occupy. Types of physical variables include measurements of watershed characteristics (i.e., watershed size, shape, land use), riparian habitat (i.e., riparian width, canopy cover, tree species composition), geomorphology (i.e., sinuosity, gradient, cross-section area), and instream habitat (i.e., wet width, water depth, water velocity, substrate types). Watershed habitat variables are typically evaluated through examination of topographic maps or use of geographic information systems to analyze electronic aerial photos, topographic maps, and other information sources. Alternatively, measurement of riparian habitat, geomorphology, and instream habitat requires field work.

Transect-based and visual-based sampling methods represent the two general approaches used for the measurement of riparian habitat, geomorphology, and instream habitat. Transect-based sampling methods consist of measuring selected habitat variables at predetermined points along multiple transects within a site. Transect-based habitat sampling methods result in quantitative habitat data, reduce observer bias, and ensure comparability among sampling sites, time periods, and categories of interest (Simonson et al., 1994; Wang et al., 1996). Transect-based sampling methods are a must when quantitative information on water depth, water velocity, riparian characteristics, channel size, and adjacent land use is needed. Visual-based habitat methods involve the estimation rather than the actual measurement of a site's habitat variables. Visual habitat methods are frequently used because they require less equipment and time than quantitative habitat measurements. Visual-based habitat methods are dependent on observer training and skills, thus scores differ among observers (Somerville and Pruitt, 2004). Visual-based habitat sampling protocols are useful for descriptive purposes and for preliminary habitat assessments intended to assist with site selection. Additionally, visual-based methods are often developed to result in the calculation of a habitat quality index, i.e., USEPA Rapid Bioassessment Habitat Index (Barbour et al., 1999), that provides an easily interpretable value representative of the physical habitat quality of the site.

The frequency of measuring physical habitat variables also depends on the goal of the surface water quality program and the type of habitat variable. For example, if the objective of the surface water project is to characterize the water chemistry conditions within the watershed and physical habitat is just being measured to provide accurate descriptions of the study sites, one-time measurements of selected habitat variables may be appropriate. However, if the objective of the surface water project is to evaluate the potential impacts of urbanization on water chemistry, geomorphology, and instream habitat, physical habitat variables may need to be measured more than once. Watershed variables, such as watershed size and shape, are not likely to change for the duration of the monitoring period and one-time measuring of these variables is appropriate. However, watershed land use, riparian, and geomorphology variables can fluctuate annually and may require annual measurements at a minimum. Many instream habitat variables exhibit seasonal and even daily fluctuations and, thus, these variables need to be measured more often than once a year. Additionally, monitoring projects with objectives to link changes in instream habitat to biological characteristics need to ensure that habitat and biological sampling events are conducted concurrently.

### 5.8.2 Biological Assessments

There are different types of biological assessments that can be used to assess stream water quality, and these different approaches are capable of assessing individual, population, or community responses. For example, one can sample animals or plants from different sites and measure the amount of contaminants within their tissues. Additionally, one could collect water from selected sites and conduct controlled laboratory bioassays that investigate the survival of laboratory animals in collected water. A common approach used in the United States and Europe is to evaluate the community characteristics of aquatic organisms from different sites (Cairns and Pratt, 1993). Also, community assessments are cheaper than conducting toxicological bioassays and in some cases cheaper than measuring water chemistry variables in the laboratory (Yoder and Rankin, 1995). We will focus on descriptions of community assessments in this section.

In general, community assessments involve the use of standardized sampling protocols to collect all species of aquatic organisms from a sampling site with the intent of calculating different metrics that describe the diversity, abundance, and species composition of the sampled communities. Sampling protocols for a wide range of stream organisms have been established by state agencies in the United

States that include algae, aquatic plants, plankton, aquatic macroinvertebrates, fish, amphibians, waterfowl, and other vertebrates (USEPA, 2002). Forty-nine of 50 states sample aquatic macroinvertebrates as part of their stream bioassessments and 35 states sample fishes (USEPA, 2002). Additionally, 33 states evaluate both aquatic macroinvertebrates and fishes as part of their stream bioassessments of water quality (USEPA, 2002).

Numerous sampling methods exist for aquatic macroinvertebrates, but the most common method used by state agencies in the United States is the dipnet (Figure 5.2).

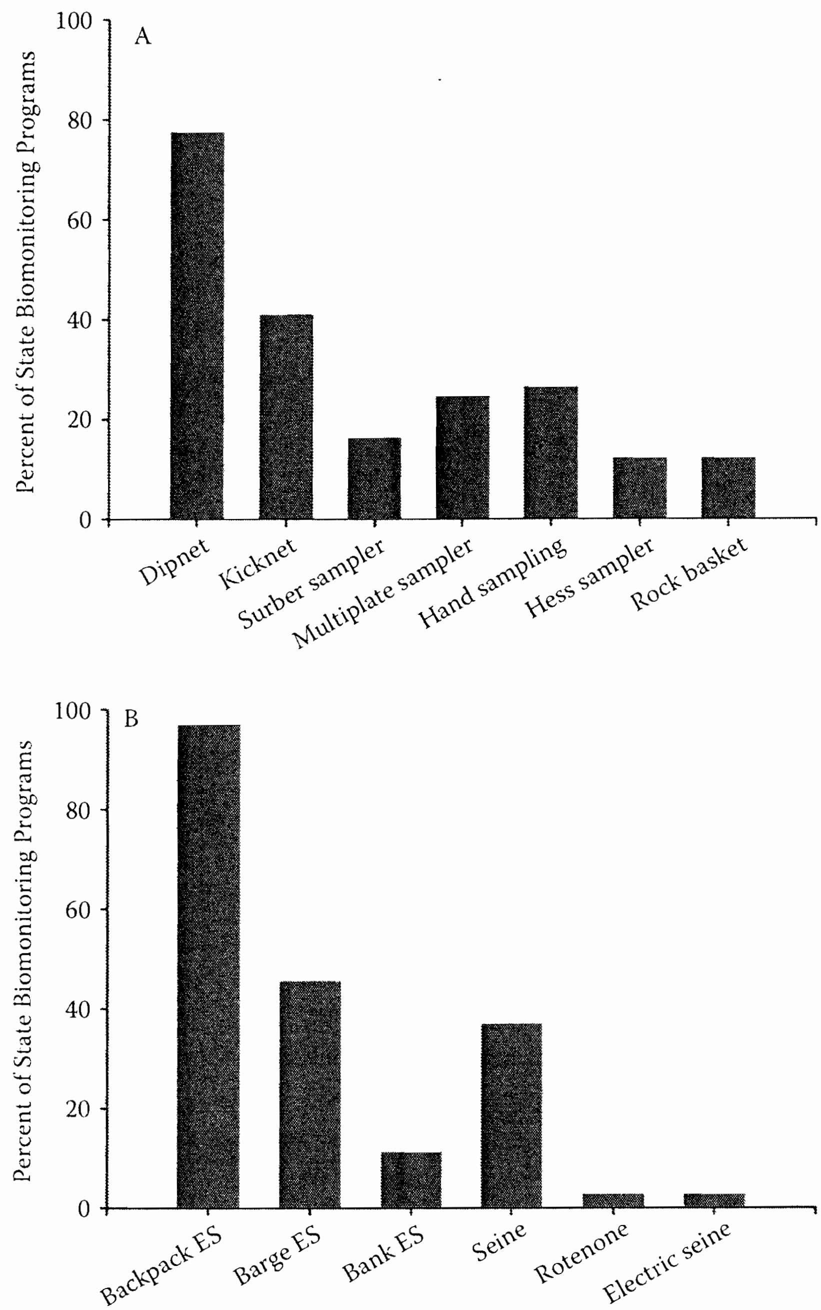


**FIGURE 5.2** Percent of use of different macroinvertebrate (A) and fish sampling techniques (B) by state biomonitoring programs in the United States (ES—electroshocker).

Essentially, this is a net with a wooden handle. The advantage of the dipnet is its capability of sampling many different types of microhabitats (i.e., pools, riffles, runs, etc.) found within a site. The disadvantage of the dipnet is that it is considered a qualitative sampling technique because one cannot calculate density (number of organisms per square meter) from dipnet samples. However, despite this flaw, the dipnet sampling methodology can be modified to standardize for sampling effort and to obtain comparable estimates of macroinvertebrate abundance. Another sampling method consists of using a surber sample, that is, a portable, stream bottom sampler consisting of two folding frames with netting that are affixed at right angles (Figure 5.3). For sampling, the horizontal frame is positioned in the substrate, and silt and rocks are stirred up so that the current transports bottom organisms into the net. The surber sampler has been most frequently used in quantitative biomonitoring studies evaluating the effects of different disturbances in gravel bottom streams (Resh and McElravy, 1993). However, the surber sampler is most effective in shallow riffles, and this explains the lack of its adoption in statewide biomonitoring programs that require multihabitat assessments of macroinvertebrate communities. Additionally, it is frequently recommended that multiple sampling techniques be used for sampling macroinvertebrates due to the diversity of the taxa involved (Karr and Chu, 1999), and many state biomonitoring programs in the United States have incorporated this recommendation (USEPA, 2002).

Capture of stream fishes typically involves direct current electrofishing; backpack-mounted electrofishers are the most commonly used in wadeable streams (Figure 5.2).

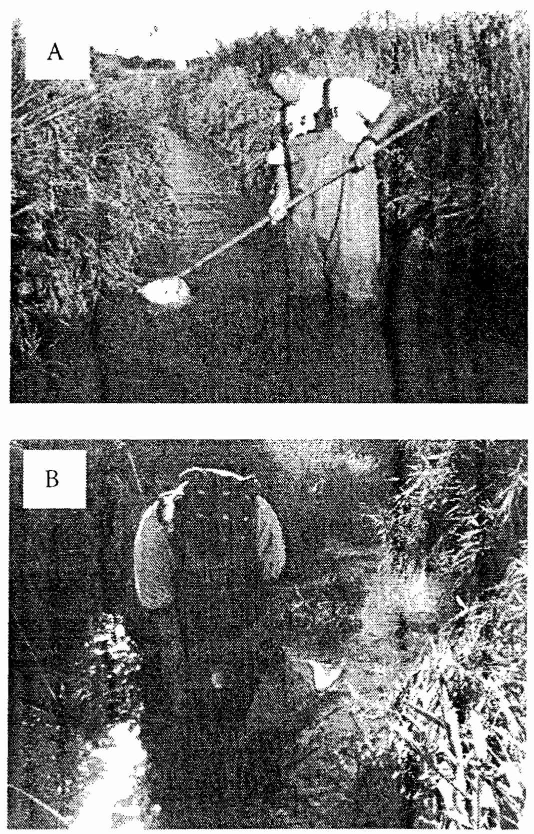


**FIGURE 5.3** Use of dipnet (A) and surber sampler (B) to collect macroinvertebrates from agricultural drainage ditches in Ohio.

Techniques for use of backpack-mounted electrofishers involve two or more people where one person carries and operates the backpack electrofisher while the other(s) dipnet the stunned fishes. The effectiveness of electrofishing has resulted in less emphasis on the use of multiple sampling techniques for fishes. However, no single sampling gear is effective in the capture of all types and sizes of organisms; thus, the use of multiple sampling techniques for fishes is also a recommended practice (Moulton et al., 2002; Smiley et al., 2009). Seining is the most frequently used non-electrofishing gear and can be a supplementary sampling method for electrofishing.

## 5.9 SUMMARY

This chapter focuses on procedures for determining chemical quality in wadeable streams and canals but also provides introductory concepts related to physical and biological aspects of water quality. Surface water sampling reflects the changing views of water quality and the emerging variety of sampling projects and goals. The first step in designing a surface water sampling project is to identify its goal and the resources available to achieve that goal. Whether the project and its goals are very simplistic or highly complex, project design should include initial planning that considers (1) the spatial extent of sampling, (2) the response variables to be analyzed, (3) the flow conditions to be sampled, and (4) the type of technology to be used. Then sampling project components (site locations, discharge measurement, sampling methodology, and personnel and equipment requirements) should be evaluated and implemented to achieve the project goal with the available resources. This chapter does not provide an exhaustive guide but rather concepts that should be integrated into a surface water quality sampling project. Readers interested in further details regarding selected topics should refer to the recommended readings section in Table 5.1.

**TABLE 5.1**
**Suggested References for Further Information on Selected Topics**

| Topic | References |
|---|---|
| Discharge measurements—theory and procedures | Brakensiek et al., 1979; Buchanan and Somers, 1976, 1982; Kennedy, 1984; Carter and Davidian, 1989; Chow et al., 1988; Haan et al., 1994; Maidment, 1993; Burton and Pitt, 2002 |
| Weirs and flumes | Bos, 1976; Brakensiek et al., 1979, USDIBR, 2001 |
| Water chemistry sampling | Dissmeyer, 1994; USDA, 1996; USEPA, 1997; Harmel et al., 2006a |
| Manual water chemistry sampling | Wells et al., 1990; USGS, 1999 |
| Water chemistry quality assurance/quality control | Dissmeyer, 1994; USDA, 1996, USEPA, 1997; Harmel et al., 2006b |
| Sediment sampling | Edwards and Glysson, 1999 |
| Storm event sampling | Wells et al., 1990; USGS, 1999; Harmel et al., 2003, 2006a |
| Habitat and biological sampling | Merritt and Cummins, 1996; Murphy and Willis, 1996; Bain and Stevenson, 1999; Barbour et al., 1999; Johnson et al., 2001; NRCS, 2001; Ohio EPA, 2002, Moulton et al., 2002; Somerville and Pruitt, 2004; Hauer and Lambertini, 2006 |

## REFERENCES

Abtew, W., and B. Powell. 2004. Water quality sampling schemes for variable-flow canals at remote sites. *J. Amer. Water Resources Assoc.* 40(5): 1197–1204.

Agouridis, C. T., and D. R. Edwards. 2003. The development of relationships between constituent concentrations and generic hydrologic variables. *Trans. ASAE* 46(2): 245–256.

Bain, M. B., and N. J. Stevenson. 1999. Aquatic habitat assessment: Common methods. American Fisheries Society, Bethesda, MD.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

Behrens, B., J. H. Riddle, and J. Gillespie. 2004. Tips to improve wet weather monitoring. *American Public Works Assoc. Reporter* Sept. 2004 (online). Also presented by K. F. Holbrook as "Maximizing reliability of storm water monitoring data" at the North American Surface Water Quality Conference, Palm Desert, Cal., July 29, 2004.

Benson, M. A., and T. Dalrymple. 1984. General Field and Office Procedures for Indirect Discharge Measurements, USGS, Washington, DC.

Bonta, J. V. 1999. Water sampler and flow measurement for runoff containing large sediment particles. *Trans. ASAE* 42(1): 107–114.

Bonta, J. V. 2002. Modification and performance of the Coshocton wheel with the modified drop-box weir. *J. Soil Water Cons.* 57(6): 364–373.

Bos, M. G., ed. 1976. Discharge measurement structures. Publication No. 20. Wageningen, The Netherlands: International Institute for Land Reclamation and Improvement.

Brakensiek, D. L., H. B. Osborn, and W. J. Rawls, coordinators. 1979. Field Manual for Research in Agricultural Hydrology. Agriculture Handbook No. 224. Washington, D.C.: USDA.

Buchanan, T. J., and W. P. Somers. 1976. Chapter A8: Discharge measurements at gaging stations. Techniques of Water-Resources Investigations of the U.S. Geological Survey, Book 3. Washington, D.C.: USGS.

Buchanan, T. J., and W. P. Somers. 1982. Chapter A7: Stage measurement at gaging stations. Techniques of Water-Resources Investigations of the U.S. Geological Survey, Book 3. Washington, D.C.: USGS.

Burton, G. A., Jr., and R. E. Pitt. 2002. *Stormwater Effects Handbook.* Boca Raton, FL: CRC Press.

Cairns, J. C., Jr., and J. R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. Pages 10–27 in D. M. Rosenberg and V. H. Resh (eds.), *Freshwater Biomonitoring and Benthic Macroinvertebrates*. Chapman and Hall.

Carey, R. O., and K. W. Migliaccio. 2009. Contribution of wastewater treatment effluents to nutrient dynamics in aquatic systems: A review. Environmental Management 44(2):205–217. DOI 10.1007/s00267-009-9309-5.

Carter, R. W., and J. Davidian. 1989. Chapter A6: General procedure for gaging streams. Techniques of Water-Resources Investigations of the U.S. Geological Survey, Book 3. Washington, D.C.: USGS.

Chow, V. T., D. R. Maidment, and L. W. Mays. 1988. *Applied Hydrology.* New York: McGraw-Hill.

Costa, J. E., R. T. Cheng, F. P. Haeni, N. Melcher, K. R. Spicer, E. Hayes, W. Plant, K. Hayes, C. Teague, and D. Barrick. 2006. Use of radars to monitor stream discharge by noncontact methods. *Water Resources Research* 42:1–14. doi:10.1029/2005WR004430.

Dissmeyer, G. E. 1994. Evaluating the effectiveness of forestry best management practices in meeting water quality goals or standards. Misc. Publication No. 1520. Atlanta, Ga.: USDA Forest Service, Southern Region.

Edwards, T. K., and G. D. Glysson. 1999. Chapter C2: Field methods for measurement of fluvial sediment. Techniques of Water-Resources Investigations of the U.S. Geological Survey, Book 3. Washington, D.C.: USGS.

Edwards, W. M., H. E. Frank, T. E. King, and D. R. Gallwitz. 1976. Runoff sampling: Coshocton vane proportional sampler. ARS-NC-50. Washington, D.C.: USDA-ARS.

Franklin, D. H., M. L. Cabrera, J. L. Steiner, D. M. Endale, and W. P. Miller. 2001. Evaluation of percent flow captured by a small in-field runoff collector. *Trans. ASAE* 44(3): 551–554.

Frissell, C. A., N. L. Poff, and M. E. Jensen. 2001. Assessment of biotic patterns in freshwater ecosystems. Pages 390–403 in M. E. Jensen and P. S. Bourgeron (eds), *A Guidebook for Integrated Ecological Assessments*. Springer: New York.

Geib, H. V. 1933. A new type of installation for measuring soil and water losses from control plots. *J. American Soc. Agron.* 25: 429–440.

Ging, P. 1999. Water-quality assessment of south-central Texas: Comparison of water quality in surface-water samples collected manually and by automated samplers. USGS Fact Sheet FS-172-99. Washington, D.C.: USGS.

Glasgow, H. B., J. M. Burkholder, R. E. Reed, A. J. Lewitus, and J. E. Kleinman. 2004. Real-time remote monitoring of water quality: A review of current applications, and advancements in sensor, telemetry, and computing technologies. *J. Experimental Marine Biol. Ecology* 300(1–2):409–448.

Gray, J. R., ed. 2005. *Proceedings of the Federal Interagency Sediment Monitoring Instrument and Analysis Research Workshop*. USGS Circular 1276. Reston, VA: USGS).

Gray, J. R., and G. D. Glysson, eds. 2003. *Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates*. [USGS Circular 1250]. Reston, VA: USGS).

Haan, C. T., B. J. Barfield, and J. C. Hayes. 1994. *Design Hydrology and Sedimentology for Small Catchments.* New York: Academic Press.

Haggard, B. E., T. S. Soerens, W. R. Green, and R. P. Richards. 2003. Using regression methods to estimate stream phosphorus loads at the Illinois River, Arkansas. *Applied Eng. Agric.* 19(2): 187–194.

Harmel, R. D., and K. W. King. 2005. Uncertainty in measured sediment and nutrient flux in runoff from small agricultural watersheds. *Trans. ASAE* 48(5): 1713–1721.

Harmel R. D., K. W. King, and R. M. Slade. 2003. Automated storm water sampling on small watersheds. *Applied Eng. Agric.* 19(6): 667–674.

Harmel, R. D., K. W. King, B. E. Haggard, D. G. Wren, and J. M. Sheridan. 2006a. Practical guidance for discharge and water quality data collection on small watersheds. *Trans. ASABE* 49(4): 937–948.

Harmel, R. D., R. J. Cooper, R. M. Slade, R. L. Haney, and J. G. Arnold. 2006b. Cumulative uncertainty in measured streamflow and water quality data for small watersheds. *Trans. ASABE* 49(3): 689–701.

Hauer, F. R., and G. A. Lamberti. 2006. *Methods in Stream Ecology*. Academic Press, Boston, MA.

Harvey, J. W., and B. J. Wagner. 2000. Chapter 1: Quantifying hydrologic interactions between streams and their subsurface hyporheic zones. In *Streams and Ground Waters*. Jones, J.B., and P.J. Mulholland., Ed. Academic Press: San Diego, CA.

Huber, W. C. 1993. Chapter 14: Contaminant transport in surface water. *Handbook of Hydrology* (ed., David R. Maidment) New York: McGraw-Hill, Inc.

Institute of Hydrology. 1980a. Low flow studies. Report No. 1. Wallingford, U.K.

Institute of Hydrology. 1980b. Low flow studies. Report No. 3. Wallingford, U.K.

Johnson, D. H., N. Pimman, E. Wilder, J. A. Sliver, R. W. Plotnikoff, B. C. Mason, K. K. Jones, P. Roger, T. A. O'Neill, and C. Barrett. 2001. Inventory and Monitoring of Salmon Habitat in the Pacific Northwest: Directory and Synthesis of Protocols for Management/Research and Volunteers in Washington, Oregon, Idaho, Montana, and British Columbia. Washington Department of Fish and Wildlife, Olympia, WA.

Karr, J. R., and E. W. Chu. 1999. *Restoring Life in Running Waters: Better Biological Monitoring*. Island Press: Washington, D.C.

Kennedy, E. J. 1984. Chapter A10: Discharge ratings at gaging stations. Techniques of Water-Resources Investigations of the U.S. Geological Survey, Book 3. Washington, D.C.: USGS.

King, K. W., and R. D. Harmel. 2003. Considerations in selecting a water quality sampling strategy. *Trans. ASAE* 46(1): 63–73.

King, K. W., R. D. Harmel, and N. R. Fausey. 2005. Development and sensitivity of a method to select time- and flow-paced storm event sampling intervals. *J. Soil Water Cons.* 60(6): 323–331.

Lazorchak, J. M., D. J. Klemm, and D.V. Peck. 1998. Environmental Monitoring and Assessment Program - Surface Waters: Field Operations and Methods for Measuring the Ecological Condition of Wadeable Streams. EPA/620/R-94/004F. U.S. Environmental Protection Agency, Washington, D.C.

Lee, K. K. 1995. Stream Velocity and Dispersion Characteristics Determined by Dye-Tracer Studies on Selected Stream Reaches in the Willamette River Basin, Oregon. U.S. Geological Survey Water-Resources Investigations Report 95-4078, 39 pp.

Lyons, J. 1992. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. *North Amer. J. Fisheries Management* 12:198–203.

Maidment, D. R., ed. 1993. *Handbook of Hydrology*. New York: McGraw-Hill.

Malone, R. W., J. V. Bonta, and D. R. Lightell. 2003. A low-cost composite water sampler for drip and stream flow. *Applied Eng. Agric.* 19(1): 59–61.

Martin, G. R., J. L. Smoot, and K. D. White. 1992. A comparison of surface-grab and cross-sectionally integrated streamwater-quality sampling methods. *Water Environ. Res.* 64(7): 866–876.

McFarland, A., and L. Hauck. 2001. Strategies for monitoring nonpoint-source runoff. TIAER Report No. 0115. Stephenville, Texas: Tarleton State University, Texas Institute for Applied Environmental Research.

Merritt, R. W., and K. W. Cummins. 1996. *An Introduction to the Aquatic Insects of North America*. Third edition. Kendall/Hunt, Dubuque, IA.

Migliaccio, K. W., B. E. Haggard, I. Chaubey, and M. D. Matlock. 2007. Water quality monitoring at the War Eagle Creek watershed, Beaver Reservoir, Arkansas. *Trans. ASABE* 50(6):2007–2016.

Miller, T. 2005. Monitoring in the 21st century to address our nation's water-resource questions. USGS Congressional Briefing Sheet. Washington, D.C.: USGS.

Morlock, S. E. 1996. Evaluation of acoustic Doppler current profiler measurement of river discharge. U.S. Geological Survey. Water Resources Investigation Report 95-4218, 37 pp.

Moulton, S. R., II, J. G. Kennen, R. M. Goldstein, and J. A. Hambrook. 2002. Revised Protocols for Sampling Algal, Invertebrate, and Fish Communities as Part of the National Water-Quality Assessment Program. Open-file report 02-150, United States Geological Survey, Reston, Virginia.

Murphy, B. R., and D. W. Willis. 1996. *Fisheries Techniques*. American Fisheries Society, Bethesda, MD.

Natural Resources Conservation Service (NRCS), 2001. Stream Corridor Inventory and Assessment Techniques: A Guide to Site, Project, and Landscape Approaches Suitable for Local Conservation Programs. Watershed Science Institute Technical Report, Natural Resources Conservation Service, Washington, D.C.

Ohio EPA. 2002. Field Evaluation Manual for Ohio's Primary Headwater Habitat Streams. Ohio EPA, Division of Surface Water, Columbus, Ohio.

Paller, M. H. 1995. Relationships among number of fish species sampled, reach length surveyed, and sampling effort in South Carolina Coastal Plain streams. *North Amer. J. Fisheries Mgmt.* 15:110–120.

Parsons, D. A. 1954. Coshocton-type runoff samplers: Laboratory investigations. SCS-TP-124. Washington, D.C.: USDA-SCS.

Parsons, D. A. 1955. Coshocton-type runoff samplers. ARS-41-2. Washington, D.C.: USDA-ARS.

Patton, T. M., W. A. Hubert, F. J. Rahel, and K. G. Gerow. 2000. Effort needed to estimate species richness in small streams on the Great Plains in Wyoming. *North Amer. J. Fisheries Mgmt.* 20: 394–398.

Pinson, W. T., D. C. Yoder, J. R. Buchanan, W. C. Wright, and J. B. Wilkerson. 2003. Design and evaluation of an improved flow divider for sampling runoff plots. *Applied Eng. Agric.* 20(4): 433–437.

Preston, S. D., V. J. Bierman, and S. E. Silliman. 1992. Impact of flow variability on error in estimation of tributary mass loads. *J. Environ. Eng.* 118(3): 402–419.

Rantz, S. E. 1982. Measurement and computation of streamflow. Vol. 1, Measurement of Stage and Discharge. U.S. Geological Survey Water Supply Paper, 2175, 284 pp.

Resh, V. H., and E. P. McElravy. 1993. Contemporary quantitative approaches to biomonitoring using benthic macroinvertebrates. Pages 159–194 in D. M. Rosenberg and V. H. Resh (eds.), *Freshwater Biomonitoring and Benthic Macroinvertebrates*. Chapman and Hall.

Sheridan, J. M., R. R. Lowrance, and H. H. Henry. 1996. Surface flow sampler for riparian studies. *Applied Eng. Agric.* 12(2): 183–188.

Shih, G., W. Abtew, and J. Obeysekera. 1994. Accuracy of nutrient runoff load calculations using time-composite sampling. *Trans. ASAE* 37(2): 419–429.

Simonson, T. D., J. Lyons, and P. D. Kanehl. 1994. Quantifying fish habitat in streams: Transect spacing, sample size, and a proposed framework. *North Amer. J. Fisheries Mgmt.* 14:607–615.

Slade, R. M. 2004. General methods, information, and sources for collecting and analyzing water-resources data. CD-ROM.

Smiley, P. C., Jr., F. D. Shields Jr., and S. S. Knight. 2009. Designing impact assessments for evaluating the ecological effects of conservation practices on streams in agricultural landscapes. *J. American Water Resources Assoc.* 45: 867–878.

Somerville, D. E., and B. A. Pruitt. 2004. Physical Stream Assessment: A Review of Selected Protocols for Use in the Clean Water Act Section 404 Program. Prepared for U. S. Environmental Protection Agency, Wetlands Division (Order # 3W-0503-NATX), Washington, D.C.

Tate, K. W., R. A. Dahlgren, M. J. Singer, B. Allen-Diaz, and E. R. Atwill. 1999. Timing, frequency of sampling affect accuracy of water-quality monitoring. *California Agric.* 53(6): 44–49.

USDA. 1996. Part 600: Introduction. In *National Water Quality Handbook*. Washington, D.C.: USDA-NRCS.

USDIBR. 2001. Water Measurement Manual. A Water Resources Technical Publication. U.S. Department of the Interior Bureau of Reclamation, Water Resources Research Laboratory. Washington, D.C. Web source: http://www.usbr.gov/pmts/hydraulics_lab/pubs/wmm/.

USEPA (U.S. Environmental Protection Agency). 1992. NPDES Storm Water Sampling Guidance Document. Office of Water. EPA 833-8-92-001. Web source: http://www.epa.gov/npdes/pubs/owm0093.pdf.

USEPA. 1997. Monitoring guidance for determining the effectiveness of nonpoint-source controls. EPA 841-B-96-004. Washington, D.C.: USEPA.

USEPA. 2002. Summary of biological assessment programs and biocriteria development for states, tribes, territories, and interstate commissions: Streams and wadeable rivers. U.S. Environmental Protection Agency, Office of Water, EPA-822-R-02-048, Washington, D.C.

USGS. 1999. Section A: National Field Manual for Collection of Water-Quality Data. Techniques of Water-Resources Investigations of the U.S. Geological Survey, Book 9. Washington, D.C.: USGS.

Vanoni, V. A., ed. 1975. *Sedimentation Engineering.* New York: ASCE.

Wang, L., T. D., Simonson, and J. Lyons. 1996. Accuracy and precision of selected stream habitat estimates. *North Amer. J. Fisheries Mgmt.* 16:340–347.

Wells, F. C., W. J. Gibbons, and M. E. Dorsey. 1990. Guidelines for collection and field analysis of water-quality samples from streams in Texas. USGS Open-File Report 90-127. Washington, D.C.: USGS.

Wilde, F. D., and D. B. Radtke. 2005. Chapter A6: Field measurements: General information and guidelines. Techniques of Water-Resources Investigations of the U.S. Geological Survey, Book 9. Washington, D.C.: USGS.

Wren, D. G., B. D. Barkdoll, R. A. Kuhnle, and R. W. Derrow. 2000. Field techniques for suspended-sediment measurement. *J. Hydraulic Eng.* 126(2): 97–104.

Yoder, C. O., and E. T. Rankin. 1995. Biological criteria program development and implementation in Ohio. Pages 109 to 144 in W. S. Davis and T. P. Simon (eds.), *Biological Assessment and Criteria.* Lewis Publishers, Boca Raton, FL.