PRACTICAL GUIDANCE FOR DISCHARGE AND WATER QUALITY DATA COLLECTION ON SMALL WATERSHEDS

R. D. Harmel, K. W. King, B. E. Haggard, D. G. Wren, J. M. Sheridan

ABSTRACT. *Many sampling projects have been initiated or modified in recent years to quantify the effects of water quality protection and enhancement programs. Although comprehensive references on the theory and procedures related to discharge data collection have been published, similar guides to water quality sampling are not available. Several sources provide general guidance on sampling project design and on manual sampling procedures, but only recently has detailed information on automated storm water quality sampling been developed. As a result, a compilation of available information on the design of water quality sampling projects is needed to support sound decision-making regarding data collection resources and procedural alternatives. Thus, the objective of this article is to compile and present practical guidance for collection of discharge and water quality constituent data at the field and small watershed scale. The guidelines included are meant to increase the likelihood of project success, specifically accurate characterization of water quality within project resource constraints. Although many considerations are involved in establishing a successful sampling project, the following recommendations are generally applicable to field and small watershed studies: (1) consider wet-weather access, travel time, equipment costs, and sample collection method in the selection of sampling site numbers and locations; (2) commit adequate resources for equipment maintenance and repair; (3) assemble a well-trained, on-call field staff able to make frequent site visits; (4) establish reliable stage-discharge relationships for accurate discharge measurement; (5) use periodic manual grab sample collection with adequate frequency to characterize baseflow water quality; (6) use flow-interval or time-interval storm sampling with adequate frequency to characterize storm water quality; and (7) use composite sampling to manage sample numbers without substantial increases in uncertainty.*

Keywords. *Agricultural runoff, Water quality sampling, Nonpoint-source pollution, Urban storm water.*

Research and progress reporting related to water quality protection is often constrained by the lack of adequate data on constituent transport from various soil and land use conditions. The need for additional data is especially apparent related to watershed modeling, which is increasingly used to guide legal, regulatory, and programmatic decision-making (Sharpley et al., 2002). The resource investment in the recently initiated USDA Conservation Effects Assessment Project (CEAP) illustrates the importance of collecting such data. In CEAP, the hydrologic and water quality effects of agricultural conservation practices are being measured at various scales (Mausbach and Dedrick, 2004). In addition to CEAP, many other federal, state, and local projects have been recently initiated or modified to quantify the storm and baseflow water quality impacts of various environmental protection and enhancement programs.

Although many such sampling projects have been implemented, practical information on state-of-the-art storm water data collection methodology has only recently been developed (e.g., McFarland and Hauck, 2001; Harmel et al., 2003; Haggard et al., 2003; Behrens et al., 2004). This recently developed information needs to be compiled and presented along with traditional methods so that project designers can make sound decisions regarding monitoring resources and procedural alternatives. Without such guidance, projects using these new methods will continue to be designed based on field experience (best case scenario) or with no regard for potential data quality implications (worst case scenario).

Therefore, the objective of this article is to provide guidance for collection of discharge and water quality constituent data at the field and small watershed scale and thus establish a practical, scientific basis for sampling project design. (In this article, discharge is synonymous to surface flow and refers to the movement of water past the location of measurement.) Methods for both discharge and water quality are described because of the direct linkage between flow and constituent transport. The well-established methods of discharge measurement and manual water quality sampling are described only briefly because comprehensive guidance is readily available from other sources. In contrast, practical guidance on allocation of data collection resources and on the advantages and disadvantages of automated and alternative sampling procedures is not currently available. These topics

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are discussed in detail and presented in the context of achieving accurate characterization of water quality within project resource constraints. Much of the guidance presented is based on practical experience of the authors, but information published in the scientific literature is included wherever possible.

The influence of scale on constituent transport is well known (Sharpley et al., 2002), but categorization of various watershed scales is difficult due to the variable nature of watershed sizes, which are determined by hydroclimatic setting and the arbitrary selection of watershed outlet locations. However, with this variability in mind, the methods discussed are generally applicable for field scale (<50 ha) to small watershed scale (<10,000 ha) data collection.

SAMPLING PROJECT DESIGN CONSIDERATIONS

This article addresses data collection related to water quality characterization at the field and small watershed scale; therefore, it focuses on methods for the determination of surface runoff and stream flow and the concentration and load of constituents associated with that discharge. Specifically, water quality in terms of nutrient and sediment transport is addressed. Related issues, such as laboratory quality assurance and quality control (QA/QC), pesticide transport and environmental effects, and aquatic ecosystem assessment, are not addressed.

The success of monitoring projects is typically determined by the tradeoff between the resources available for data collection and accurate characterization of water quality conditions. The importance of this tradeoff is illustrated by research on the subject (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). The issue of optimal allocation of project resources such that water quality is accurately characterized within resource constraints should underlie each decision in project design (Abtew and Powell, 2004; Miller, 2005). Achieving an appropriate balance requires careful decision-making on the type, amount, and quality of data collected. Information on the design considerations discussed in the following sections (data collection resources, discharge characterization, water quality characterization, automated storm sampling design components, and alternative sampling procedures) is required to make sound decisions.

DATA COLLECTION RESOURCES

Most projects designed to collect discharge and water quality data are constrained by limited resources (Harmel et al., 2003; Abtew and Powell, 2004; Miller, 2005). Agouridis and Edwards (2003) emphasized that the collection and analysis of water quality samples is difficult, time consuming, and expensive. As projects are designed or modified, monitoring resources should be carefully allocated between site establishment, equipment purchase and maintenance, personnel requirements, and sample analysis. If sampling resources are inappropriately allocated, the quality of collected data will suffer.

Site Establishment

Decisions related to the number and location of sampling sites directly affect monitoring resources (USDA, 1996). Each additional sampling site increases equipment and sample analysis costs and personnel travel time. The location of sampling sites should be selected to minimize travel time, if possible (USDA, 1996). Long trips to distant sampling sites can be especially difficult and costly in wet periods when frequent trips are required.

Field-scale sampling sites should be established at the boundaries of homogeneous land use areas, preferably within the natural drainage way (USDA, 1996), but berm construction may be necessary to direct runoff to a single outlet for each field. Small watershed-scale sampling sites are necessary to determine integrated effects of upstream conditions on water quality. To adequately characterize the water quality issues of interest, the location and influence of constituent sources such as wastewater treatment plants, construction sites, and stream modification should be examined when locating downstream sites. Where possible, sites should be located 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 sites and stage-discharge relationships in morphologically active channels (discussed in detail in subsequent sections).

Shelters should be built to house and protect data collection equipment at all sampling sites. The shelters should be located above the highest expected flow elevation and be accessible during high flows (Haan et al., 1994; USEPA, 1997). Livestock, rodents, and insects can damage equipment and contaminate samples, so they should be controlled in and around equipment shelters, electric lines, communication cables, and sample tubes.

Equipment Purchase and Maintenance

Purchase of data collection equipment for each site requires a substantial initial investment. In addition, purchase of duplicate, backup equipment to substitute for malfunctioning components is recommended. Automated samplers are especially expensive compared to the alternative of manual sampling but can improve data quality in many situations and decrease safety risks. Similarly, pre-calibrated weirs and flumes are expensive to purchase and install but can improve flow measurement if correctly installed and operated. Detailed information regarding the advantages and disadvantages of these alternatives is presented in subsequent sections. Stage measurement devices are also quite expensive but generally necessary.

In spite of the required expense and time commitment, equipment maintenance must remain a high priority to ensure meaningful data (USEPA, 1997). A commitment to proper maintenance limits data loss and equipment malfunctions, which if allowed to occur increase the uncertainty in measured data (USDA, 1996). Maintenance visits to each sampling site, whether remote or readily accessible, should be made weekly or biweekly to:

- Inspect power sources, stage recorders, pumps, sample tubes, sample intakes, and dessicant strength.
- Calibrate stage recorders to ensure flow measurement accuracy.
- Retrieve collected data to limit the amount of data lost in potential power failures or other malfunctions.

Personnel Requirements

Committed, on-call field staff are essential to successful water quality sampling projects. Field personnel should be well-trained on QA/QC methodology, equipment operation, basic hydrology, and safety considerations (USEPA, 1997). Whether samples are collected manually or automatically, personnel must make frequent trips to sampling sites to collect data and retrieve water samples. In either case, field staff must also commit adequate time to conduct necessary equipment inspection, maintenance, and repair. These site trips are often necessary with little advance warning and under adverse weather conditions, especially for manual sample collection. If samples are collected automatically, personnel should visit all sampling sites as soon as possible (as determined by QA/QC guidelines) after sampling events to retrieve samples, inspect flow measurement and automated sampler function, and make necessary repairs. Excessive delay in retrieving water samples can result in changes to their chemical composition and thus inaccurate representation of actual water quality.

Sample Analysis

It is also important to be mindful of the number of samples that can be collected and analyzed by a laboratory in a reasonable time frame, as determined by QA/QC guidelines (USDA, 1996), and remain within the laboratory analysis budget (Novotny and Olem, 1994). It is prudent to estimate the number of samples that will be collected (Harmel et al., 2003) so that reasonable sampling expectations can be set. Many of the project design considerations discussed subsequently affect the number of samples collected, which directly influences sample analysis costs.

Discharge Characterization

Collection of appropriate surface flow data is essential to adequately characterize water quality on small watersheds because discharge processes determine: (1) offsite constituent loss, (2) downstream constituent transport, and (3) channel erosion and deposition. In addition, discharge data along with associated constituent concentrations are needed to determine mass transport (load values) and to differentiate between transport mechanisms.

Discharge Measurement

Much of the information on accepted methods in streamflow data collection compiled for this article was developed by USDA and USGS scientists and appears in the Field Manual for Research in Agricultural Hydrology (Brakensiek et al., 1979) and in selected Techniques of Water-Resources Investigations of the U.S. Geological Survey (e.g., Buchanan and Somers, 1976, 1982; Kennedy, 1984; Carter and Davidian, 1989). Chow et al. (1988), Haan et al. (1994), and Maidment (1993) are additional comprehensive sources that provide valuable information on applied hydrology. These well-established methods are described only briefly because comprehensive guidance is readily available.

The most common continuous discharge measurement method utilizes stage (water surface level or flow depth), which can be readily measured with a variety of devices, and its relationship to discharge. With this method, a stage-discharge relationship (rating curve) is established for the site, and then stage data are recorded and translated to discharge with that relationship. A general description of stage-discharge relationships and their development is provided in most applied hydrology texts (e.g., Brakensiek et al., 1979; Maidment, 1993) and in selected Techniques of Water-Resources Investigations of the U.S. Geological Survey (e.g., Buchanan and Somers, 1976, 1982; Kennedy, 1984; Carter and Davidian, 1989).

The stage-discharge relationship either accompanies pre-calibrated structures, such as flumes or weirs, or must be developed with a series of stage and discharge measurements. For small watershed sites, pre-calibrated flow control structures are highly recommended because they have an associated stage-discharge relationship and provide reliable and accurate flow data (Slade, 2004) for a number of years with minimal maintenance. Although weirs or flumes are highly recommended, they are expensive to purchase and install and can cause flow ponding, which can impact sediment transport (USDA, 1996). To select an appropriate structure for site-specific flow and constituent transport conditions, the following factors should be considered: (1) range of flows and its effect on hydraulic performance and measurement accuracy, (2) existing headwater-tailwater influences on the applicability of structure calibration (limits of modularity), (3) presence of floating or suspended debris and transported sediment, (4) costs of construction and maintenance, (5) expected life of the project, and (6) potential benefits of flow measurement standardization within the project. In settings with extreme sediment loads, sediment deposition can invalidate the stage-discharge relationship of typical pre-calibrated structures. In such settings, a drop-box weir can be used to accurately measure discharge (Brakensiek et al., 1979; Bonta and Pierson, 2003). The drop-box weir creates turbulence that suspends sediment and bedload while obtaining a valid discharge record. Recent investigations of the hydraulic performance of the drop-box weir has extended its utility to steep slopes and skewed weir-approach channels for large and small watersheds and erosion plots (Bonta, 1998; Bonta and Goyal, 2001; Bonta and Pierson, 2003). Criteria for selection of the flow structures are examined by Bos (1976) in a comprehensive text detailing the many types of weirs and flumes available and by Brakensiek et al. (1979).

Pre-calibrated structures can, however, be limited in the discharge they support, which limits their use as the watershed scale increases. If expected flow rates exceed structure discharge capacities or if purchase and installation are not feasible, sampling sites should be located at or near established gauge stations with existing discharge data and an established stage-discharge relationship. If no such site exists, recommended sampling locations include stable channels (e.g., concrete or bedrock) with a natural flow control (e.g., rock riffle or fall) or an artificial control (e.g., low water dam), which can produce a stable stage-discharge relationship (Carter and Davidian, 1989; USDA, 1996). In contrast, it is difficult to develop and maintain reliable stage-discharge relationships at sites in morphologically active channels.

To use the stage-discharge method at sites with an uncalibrated flume or weir or without an established stage-discharge relationship, the relationship must be developed. Detailed information on developing stage-discharge relationships and choosing appropriate locations in natural channels is provided by many sources (e.g., Buchanan and Somers, 1976; Kennedy, 1984; Carter and Davidian, 1989).

Developing a stage-discharge relationship is a time consuming, long-term task. The commonly used area-velocity method requires measurement of stage, cross-sectional area, and flow velocity for a range of stages. Several portable devices are available to measure flow velocities. Velocity 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. Whichever device is used, the mean flow velocity within the section(s) of interest must be determined. In the area-velocity method, flow is divided into vertical sections, and mean velocity and cross-sectional area are determined for each section. The total discharge at that stage is the sum of discharges for each section. This procedure must be repeated for the entire range of expected discharges and should be checked periodically to determine if shifts in the relationship have occurred.

With an established stage-discharge relationship, a continuous record of stage is measured and translated to discharge. Sensor types commonly used to provide continuous stage data include bubblers, pressure transducers, non-contact sensors, and floats (Buchanan and Somers, 1982; USDA, 1996). Bubblers and pressure transducers are submerged sensors that measure stage by sensing the pressure head created by water depth. Non-contact sensors, which are suspended above the water surface, use ultrasonic or radar technology to measure water level. Each of these sensors is generally used in connection with an electronic data logger to store a continuous stage record (USDA, 1996). Float sensors actually float on the water surface, and in conjunction with a stage recorder, produce a graphical or electronic record of stage. Stage sensors should be installed in a stilling well for protection and creation of a uniform water surface for improved measurement accuracy. Installation of a permanent staff gauge is also recommended (USDA, 1996); but at minimum, a surveyed reference elevation point should be established with which to calibrate stage sensors (Brakensiek et al., 1979; Haan et al., 1994).

Other methods of discharge determination utilize more recent technology and/or introduce more uncertainty in measured discharge. One alternative utilizes permanent in-stream velocity meters and stage sensors to provide continuous measurement of flow velocity and stage. In theory, these instruments use velocity measurements and corresponding stage data with cross-sectional survey data to produce the cross-sectional flow area and determine discharge; however, reported flow velocity values may not adequately represent the mean velocity of the entire flow cross-section.

Another alternative utilizes Manning's equation to estimate discharge (Maidment, 1993; Haan et al., 1994). In this method, Manning's equation estimates flow velocity based on channel roughness, slope, and cross-sectional geometry. Then, cross-sectional survey data are used with the velocity estimate to determine discharge. This method, however, introduces substantial uncertainty into discharge data because 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 final alternative for estimation of continuous discharge data.

WATER QUALITY CHARACTERIZATION

Several publications provide valuable information on various aspects of water quality data collection. The USGS presents its preferred methods for water quality sampling in the National Field Manual for Collection of Water Quality Data (USGS, 1999). While the USGS manual is a comprehensive guide to manual water quality sampling procedures, it does not address automated sampling techniques or monitoring resource considerations. Other publications provide extensive guidance on manual field measurements (e.g., Wells et al., 1990) and general information on QA/QC, sample collection, and statistical analysis (e.g., Dissmeyer, 1994; USDA, 1996; USEPA, 1997). The following section briefly describes this guidance but focuses on automated sampling and alternative procedures and on the accuracy of collected data.

Periodic Baseflow Sampling

Periodic baseflow sampling is necessary at intermittent and perennial flow sites to quantify the contributions of point sources, tile drainage, shallow surface return flow, and constituent release from in-stream processes. In contrast, baseflow sampling is often unnecessary at field-scale sites, which are typically characterized by ephemeral flow and constituent transport. To provide the most useful data, baseflow water quality samples should be taken as often as possible at regular time intervals. Preferably, samples should be collected weekly to better capture concentration variability, but less frequent sampling (not less than once per month) can be adequate for watersheds with increased attenuation. Typically, samples can be taken at a single point in the flow, usually in the centroid of flow, because dissolved constituent concentrations are assumed to be uniform across the cross-section, unless the site is located immediately downstream of a significant point-source contribution (Martin et al., 1992; Ging, 1999; R. Slade, personal communication, 2004). Minimal particulate matter is transported at low flow velocities, except in turbid systems, so many baseflow samples are analyzed only for dissolved constituents and pathogens.

Storm Sampling

Storm sampling is needed to quantify constituent transport in runoff events and to differentiate between various processes such as channel, point source, and nonpoint source. Storm flow transports both recently washed off and resuspended constituents that have been attenuated by in-stream processes. Characterization of storm water quality is much more difficult than baseflow characterization. Runoff events often occur with little advance warning, outside conventional work hours, and under adverse weather conditions (USEPA, 1997). As a result, small watershed projects typically utilize automated water quality sampling equipment so that personnel are not forced to travel to multiple sites during runoff events and manually collect samples under hazardous conditions. Automated samplers are extensively used because typical projects do not have the resources to maintain an adequate on-call field staff to perform manual storm sampling. (The U.S. Geologic Survey, however, is one agency with the expertise and personnel to conduct proper manual storm sampling.) Major advantages of automated samplers are their ability to use a consistent sampling procedure at multiple sites and to take multiple samples throughout entire runoff durations (table 1). Automated samplers are also able to sample within the quick hydrologic response time of small watersheds. Automated samplers are, however, expensive to purchase and maintain and thus

Table 1. Potential advantages and disadvantages of automated and manual storm sampling.

Automated Storm Sampling		Manual (EWI or EDI) Storm Sampling	
Advantages	Disadvantages	Advantages	Disadvantages
Reduced on-call travel.	Large investment in equipment.	Low equipment cost.	Frequent on-call travel often in adverse weather and dangerous conditions.
Multiple samples collected automatically.	Single sample intake (samples taken at one point in the flow).	Integrated samples throughout vertical profile and cross-section.	Time-consuming travel and sample collection make numerous sites difficult to manage.
Minimizes work in dangerous conditions.	Difficult to secure intake in centroid of flow.		Difficult to obtain samples throughout hydrograph.
Numerous sites feasible.			Large investment in personnel.

require considerable financial investment (USDA, 1996). Although these samplers automatically collect samples during storm events, they are far from trouble-free and require considerable maintenance and repair effort. Whether manual or automated storm sampling is conducted, water quality samples should be collected throughout the duration of runoff events, including flow recession.

In contrast, manual storm sampling requires personnel available to travel to each sampling site and manually collect samples during runoff events (table 1). Manual techniques require substantial collection time for each sample, making it difficult to collect multiple samples at numerous sites. Proper manual storm sampling typically utilizes the USGS equal-width increment (EWI) or equal-discharge increment (EDI) procedures (Wells et al., 1990; USGS, 1999). With these procedures, multiple depth-integrated, flow-proportional samples are obtained across the stream cross-section and produce accurate concentration measurements even in large streams, which is an important advantage. However, sample collection throughout the range of observed discharges can be difficult with these procedures. Wells et al. (1990) and USGS (1999) provide extensive guidance on manual sample collection techniques and on proper QA/QC methodology. A less-intensive manual sampling alternative, grab sampling at a single collection point at random times during storm events, is less hazardous and time consuming but not recommended because it does not capture within-channel and temporal concentration variability.

Perhaps the most important difference between automated and manual storm sampling is that automated samplers typically utilize a single intake (single sampling location), but EWI and EDI procedures collect integrated samples across the flow cross-section. Thus, the uniformity of water quality across the cross-section and within the water profile deserves consideration. It is generally assumed that dissolved constituents can be adequately sampled at a single intake point at field and small stream sites because of well-mixed conditions and shallow flow depths and in larger streams unless immediately downstream of significant point sources (Martin et al., 1992; Ging, 1999; R. Slade, personal communication, 2004). This assumption can be evaluated with four parameter probe (pH, temperature, conductivity, and dissolved oxygen) measurements throughout a stream cross-section. The USGS recommends that if these 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).

In contrast, sediment and sediment-bound constituent concentrations often vary within the flow profile and across the channel. At field-scale sites and small streams, a single sample intake is generally adequate for sediment sampling because of well-mixed conditions and shallow flows; however, integrated samples are needed for coarse sediment and in larger streams to adequately capture sediment concentration variability. For larger streams, samples can be appropriately collected with the manual EWI or EDI approaches or with automated samplers supplemented by manual sampling. To use automated samplers to determine sediment and sediment-associated constituent concentrations, the relationship must be established between concentrations at the sampler intake and the total concentration as determined by integrated samples at a range of discharges (e.g., Ging, 1999). Then, concentrations at the intake can be corrected to represent those in the total cross-section (R. Slade, personal communication, 2005).

DESIGN COMPONENTS FOR AUTOMATED STORM SAMPLING

Most commercially available automated water quality samplers contain the following components: programmable electronic operation and memory, stage recorder, sample collection pump, and sample bottles (typical arrangements allow 1 to 24 sample bottles). Three main design components are critical in programming and operating automated samplers (USDA, 1996) because they determine how, how many, and when samples are taken and ultimately determine the quality of collected storm water quality data. These critical components: (1) threshold to start and finish sampling, (2) interval on which to collect samples, and (3) decision to take discrete or composite samples, are discussed in the following sections.

Storm Sampling Threshold

The first critical component in programming automated samplers is selecting a threshold at which to initiate sampling. Generally, a minimum stage or discharge threshold is set for small watershed studies. When flow exceeds this minimum flow threshold, sampling begins and continues as long as the flow remains above this threshold or until flow ceases; therefore, the sampling threshold directly affects the number of samples that are taken and the proportion of the runoff event that is sampled (fig. 1).

Results from Harmel et al. (2002) suggest that substantial sampling error is introduced as storm sampling thresholds are increased. Therefore, thresholds should be set to sample as much of the storm duration as possible. To prevent pump malfunction, ensure that the sampler intake is completely submerged at the storm sampling threshold. Ideally, the sampler intake should be located in well-mixed flow either in the center of the channel in a run/riffle, not a pool, or immediately upstream below the crest elevation of the hydraulic control structure. The programming option of collecting a sample each time flow rises and/or falls past the threshold should be avoided because flow fluctuation near the threshold can result in excessive (and unnecessary) samples.

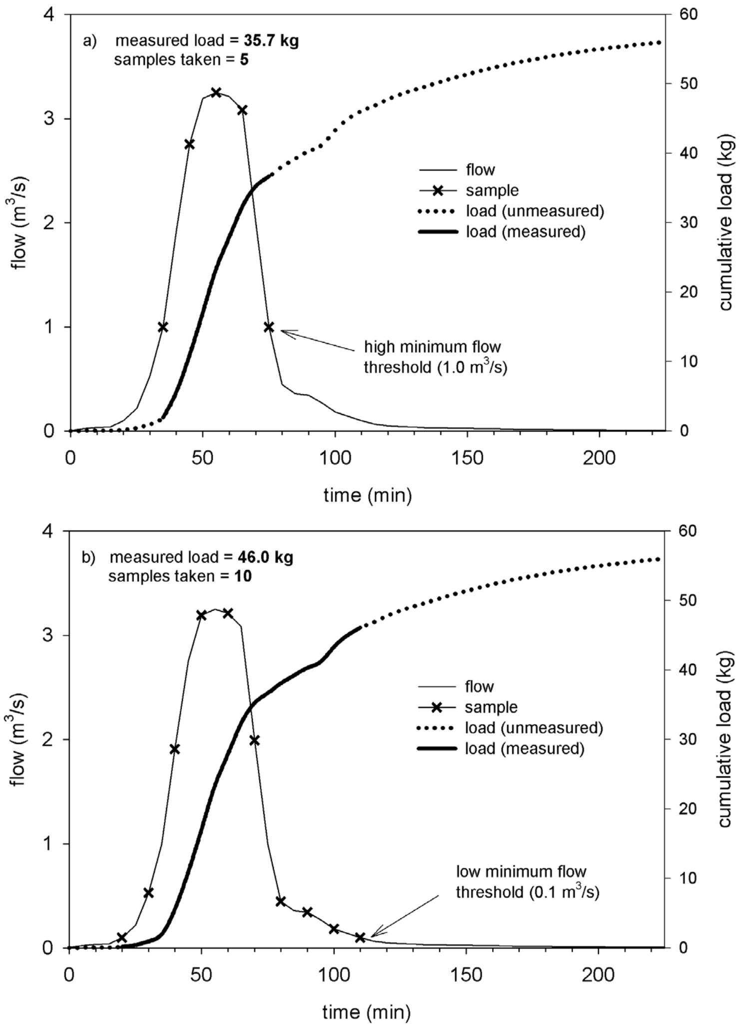

Figure 1. Hypothetical examples illustrating potential differences in measured loads for differing minimum flow thresholds. The examples are for time-interval sampling (10 min) with minimum flow thresholds of (a) 1.0 $m^3 s^{-1}$ and (b) 0.1 $m^3 s^{-1}$.

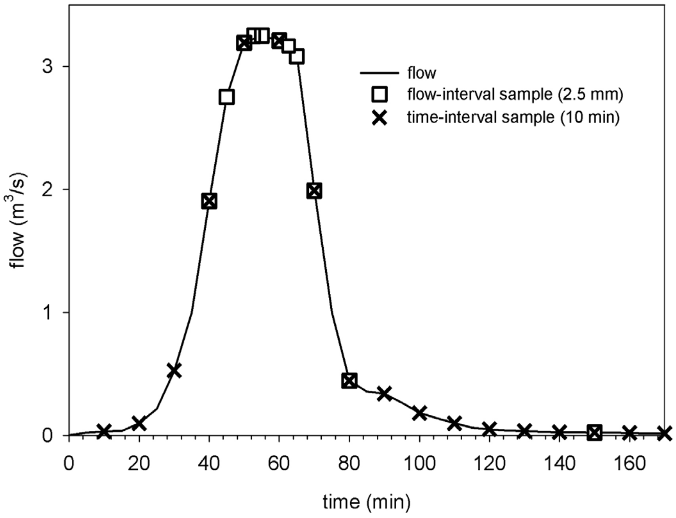

Figure 2. Illustration of differences in sample timing based on time intervals or flow intervals.

Sampling Interval

Another critical component is the interval at which to sample once the storm sampling threshold is reached. Two options are available for determining the sampling interval: time and flow (fig. 2). With time-interval sampling (also referred to as time-weighted, time-proportional, or fixed frequency sampling), samples are taken on time increments, such as every 30 min. With adequate knowledge of site hydrology on which to base time-intervals, time intervals can instead be programmed to vary (typically with more frequent samples initially, then less frequently as the storm proceeds). Time-interval sampling is a simple and reliable procedure since accurate time intervals are easy to measure and clock failures are rare. However, if small time intervals are used, frequent sampling can produce numerous samples and quickly reach the sampler capacity, thus missing a majority of runoff event. Time-interval sampling does not eliminate the need for flow measurement, as flow data are necessary for load determination.

With flow-interval sampling (also referred to as flow-weighted or flow-proportional sampling), samples are collected on flow volume increments, such as every 2000 m^3 or 2.5 mm volumetric depth. (Referring to discharge intervals in volumetric depth units such as mm, which represents mean runoff depth over the entire watershed, as opposed to volume units such as m^3, normalizes discharge over various watershed sizes. This notation allows a consistent transfer of methods and results to small watersheds of differing sizes.) Flow-interval sampling requires continuous discharge measurement to determine sampling intervals. Flow-interval sampling readily produces the event mean concentration (EMC), a common method for reporting constituent concentrations defined as the arithmetic mean of individual sample concentrations collected on equal discharge (flow-weighted) intervals. The EMC multiplied by the total flow volume represents the constituent load.

Several studies have concluded that flow-interval sampling better represents storm loads than time-interval sampling because a greater proportion of flow-interval samples are taken at higher flows with corresponding higher transport (Claridge, 1975; Richards and Holloway, 1987; Rekolainen et al., 1991; Miller et al., 2000; McFarland and Hauck, 2001; King and Harmel, 2003; Abtew and Powell, 2004; Harmel and King, 2005). Stone et al. (2000) concluded that flow-interval sampling is more accurate only when constituent concentrations are positively correlated to flow rate. In contrast, Izuno et al. (1998) and Shih et al. (1994) concluded that time- and flow-interval sampling can perform equally well in representing agricultural drainage water quality.

Statistical sampling theory indicates that the smaller the sampling interval (the more samples taken), the better actual population characteristics are estimated (Haan, 2002). Several recent studies confirm this theory regarding storm sampling (Richards and Holloway, 1987; Shih et al., 1994; Miller et al., 2000; King and Harmel, 2003, 2004; Harmel and King, 2005); thus, small sampling intervals should be used to reduce the uncertainty in water quality measurements. However, intervals must also be set to sample throughout runoff events of various durations to capture the various transport phenomena from first flush and lateral subsurface return flow. King and Harmel (2003) and Harmel et al. (2003) provide guidance on selecting time and flow intervals for automated sampling on small watersheds, and King et al. (2005) have recently developed a procedure to determine sampling intervals based on watershed and constituent characteristics.

In practical terms, it is difficult to choose time intervals that are able to completely sample events of various durations with adequate frequency to capture constituent concentration behavior without exceeding sampler capacity, which is often 24 discrete samples. In contrast, it is much easier for flow-interval sampling to intensively sample throughout events of various magnitudes (table 2). Appropriate time intervals vary considerably based on watershed characteristics, but flow intervals within a small range (1 to 6 mm) are widely applicable to small watersheds. Flow intervals up to 6 mm are appropriate for constituents that vary relatively little within runoff events, but smaller intervals (1 to 3 mm) should be used when concentrations vary widely (Harmel and King, 2005). Whichever method is used to determine sampling intervals, composite sampling can be a powerful option to increase sampling capacity.

Discrete Versus Composite Sample Collection

Automated samplers typically have the option of collecting discrete samples (one sample per bottle) or composite samples (more than one sample aliquot per bottle). Discrete sampling strategies provide the best representation of temporal variability of constituent concentrations; however, discrete sampling can produce substantial uncertainty even with small sampling intervals. This increased uncertainty is most pronounced in large-volume and/or long-duration runoff events, when sampler capacity is exceeded prior to the end of storm runoff. As shown in table 2, excessive samples are possible especially for time-interval sampling, but the common 24-bottle limitation allows only a fraction of the samples to be collected. Composite sampling increases sampler capacity by collecting more than one sample aliquot in each sample bottle, which makes it a valuable alternative. Composite sampling with two, three, or four sample aliquots per bottle reduces sample numbers to 50%, 33%, and 25% of that collected by discrete strategies. Composite sampling does, however, reduce information on the distribution of within-event constituent behavior, which limits the study of various transport mechanisms (McFarland and Hauck, 2001). For composite sampling, sample aliquot volumes should be at least 100 to 200 mL because of the difficulty in accurately pumping small volumes.

An alternative to collecting composite samples in the field involves manually compositing samples in the lab. For discrete flow-interval samples, equal-volume subsamples can be withdrawn and combined to create composite samples. For discrete time-interval samples, subsample

Table 2. The number of storm samples collected with various time-and flow-interval discrete sampling strategies for 300 runoff events examined by King and Harmel (2003); watershed sizes ranged from 0.1 to 6300 ha.

	Number of Samples Collected		
	Range	Mean	Median
Time-interval discrete sampling strategies			
5 min	8 – 1237	234	164
10 min	4 – 619	117	82
15 min	3 – 413	78	55
30 min	2 – 207	39	28
60 min	0 – 104	20	14
120 min	0 – 52	10	7
180 min	0 – 35	6	5
360 min	0 – 18	3	3
Flow-interval discrete sampling strategies			
1.0 mm	0 – 132	30	25
2.5 mm	0 – 53	12	10
5.0 mm	0 – 26	6	5
7.5 mm	0 – 17	3	3
10.0 mm	0 – 13	2	2

volumes proportional to the flow during each time interval can be withdrawn and combined to create composite samples. These manual techniques produce valid flow-weighted concentration estimates but require considerable post-processing.

Several recent studies have concluded that composite sampling introduces less error than increasing minimum flow thresholds or increasing sampling intervals, especially for flow-interval sampling (Miller et al., 2000; Harmel et al., 2002; King and Harmel, 2003; Harmel and King, 2005). Therefore, composite sampling is recommended to control the number of samples collected. For sampling projects whose primary goal is load determination, such as Total Maximum Daily Load (TMDL) projects that typically are not interested in within-event constituent behavior, single-bottle, composite flow-interval sampling is a powerful option that reduces analysis costs while intensively sampling entire event durations (Shih et al., 1994; Harmel and King, 2003). With this strategy, 80 to 160 flow-interval samples of 100 to 200 mL can be composited into a single sample (16 L bottle capacity) to produce the EMC. Another alternative is to collect discrete samples for 1 to 2 years to gain information on constituent behavior within storm events and seasonal/annual cycles and then convert to composite sampling.

ALTERNATIVE SAMPLING PROCEDURES

Mechanical Flow-Proportional Samplers

Mechanical samplers can be practical alternatives to electronic automated samplers (described in previous sections). Two of these mechanical approaches, the rotating slot sampler and the multi-slot divisor sampler, have proven useful in specific water quality applications. One advantage of these mechanical samplers is their ability to collect flow-weighted samples and estimate flow volume, and thus easily calculate EMC and mass loads.

The Coshocton wheel sampler, a rotating slot sampler developed by Pomerene (Parsons, 1954, 1955), requires limited maintenance and no electric power and provides a single flow-proportional sample of runoff from plots or small watersheds (Edwards et al., 1976). Bonta (2002) modified the original Coshocton design for sampling sediment-laden flows originating from the drop-box weir. Additionally, Bonta (1999) developed a specialzed sampler for the drop-box weir to sample flows with large sediment particles. Malone et al. (2003) modified the rotating slot design to provide a flow-proportional sample for flows ranging from slow drips to continuous flows from tile drains, springs, or lysimeters.

Variations of the multi-slot divisor developed by Geib (1933) may be useful for placement above and below individual BMPs such as riparian buffers. This type of sampler stores a known fraction of shallow overland flow and permits unattended flow-proportional sampling at multiple locations within study areas. The device is suited to locations with shallow, non-concentrated surface runoff and where minimum disturbance of soil and vegetative cover is essential. The LIFE sampler, a low-cost variation of the stationary multi-slot divisor, was developed by Sheridan et al. (1996) to meet sampling and field data needs for riparian buffer studies in regions of low slope. Franklin et al. (2001) modified the sampler for use in water quality studies on pastures sites with greater slopes. Eisenhauer et al. (2002) developed a simple, low-cost divisor for sampling overland or surface flows within riparian buffer strips. Their device retains a composite water quality sample and permits determination of the total runoff volume and re-creation of the runoff hydrograph. Another variation that utilizes a notched flow divisor was developed by Pinson et al. (2003) to measure runoff volume and water quality on study plots on steep terrain.

Regression Method

Alternative approaches to quantify water quality constituent loads utilize regression methods (Cohn et al., 1989; Cohn, 1995). In their simplest form, regression methods utilize the relation between discharge and constituent concentration and a logarithmic transformation, but they have been modified to account for nonlinearities, seasonal and long-term concentration variability, censored concentration data, and biases associated with logarithmic transformations (Cohn, 1995; Robertson and Roerish, 1999). The statistical relation between discharge, concentrations, and other complicating factors is used to estimate missing daily constituent loads, which are then summed to produce monthly, seasonal, or annual loads. A benefit of this statistical approach is its ability to place confidence limits on resulting load estimates.

Regression methods require somewhat less-intensive sampling than automated sampling and can be applied to relatively small datasets collected over many years. Sampling design to support regression methods can vary based on the duration of the study and desired load estimation period (Robertson, 2003), but it must adequately describe the relation between discharge and constituent concentration throughout the range of discharge observed at that location. Sampling strategies should target both baseflow and storm events, as fixed-interval sampling (e.g., monthly sampling) may not adequately represent the range of discharge. Monthly sampling strategies targeting baseflow may underestimate constituent loads by more than 40% (Haggard et al., 2003). Robertson and Roerish (1999) suggested that the collection of water samples during storm events may positively bias annual load estimates in smaller streams because storm concentrations are typically larger than average daily concentrations. Constituents associated with sediment transport often exhibit hysteresis within storm events, with greater concentrations on the rising portion of the hydrograph than the corresponding discharge on the falling portion (Richards and Holloway, 1987; Thomas, 1988; Richards et al., 2001); therefore, samples should be collected during both the rising and falling portions.

Regression methods have been widely used, particularly by the USGS in relatively large streams and rivers across the U.S. (e.g., Green and Haggard, 2001; Pickup et al., 2003). At that scale, automated storm sampling with a single intake may not adequately represent the average constituent concentration across the cross-section; thus, regression methods can be an effective and economical alternative to estimate constituent load transport from larger watersheds. Application of regression methods to field- and farm-scale watersheds, however, requires additional investigation because these methods can be relatively imprecise. For example, median absolute errors were ~30% in small watersheds according to Robertson and Roerish (1999).

Sediment Sampling

Because of the complexity of sediment transport measurement, especially as scale increases, a brief discussion of alternative methods for sediment load determination is warranted. Whereas automated samplers are appropriate for quantification of suspended sediment and associated constituent losses at small scales, alternative methods are often necessary at scales where channel processes become significant.

The measurement of sediment load is a difficult task because of the temporal and spatial variability in sediment transport. In particular, the size of particles in a channel is an important consideration. Particles that are <62 μm in diameter are generally homogeneously distributed throughout a channel's cross-section, making the use of automatic pumping samplers appropriate for collecting representative samples of the water/sediment mixture (Vanoni, 1975; Edwards and Glysson, 1999). However, if the particles are >62 μm, they are not homogeneously distributed with depth. A vertical gradient will be present; the concentration of particles will be much higher near the bed and decrease with increasing distance from the bed. This makes determination of the exact location of a sample with respect to the channel bed crucial. Ideally, sand-sized (62 to 2000 μm) particles should be sampled isokinetically. In isokinetic sampling, the water-sediment mixture is withdrawn at the ambient velocity of the streamline from which the sample is taken. If the sample is withdrawn too quickly, it will be enriched water because water from adjoining streamlines will be taken in while, because of their momentum, particles from the adjoining streamlines will not. If the sample is withdrawn too slowly, it will be enriched in particles because fluid will be forced to flow around the nozzle while particles, because of their increased momentum relative to water, will flow into the nozzle (ICWR Subcommittee on Sedimentation, 1963).

Particularly in the case of sampling particles >62 μm, methods established by the USGS in Field Methods for Measurement of Fluvial Sediment (Edwards and Glysson, 1999) are recommended. This work is probably the most recognized standard for sediment sampling and contains detailed descriptions of sediment sampling techniques and equipment. For the analysis of suspended sediment samples, the suspended sediment concentration (SSC) analysis method is the accepted standard (ASTM, 1999; Gray et al., 2000; Glysson et al., 2001). Various isokinetic samplers have been developed and evaluated by the Federal Interagency Sedimentation Project (FISP). More information on FISP and available samplers is available at: http://fisp.wes.army.mil. Several alternative techniques, such as acoustic backscatter and optical backscatter, for measuring suspended sediment concentrations that are not described in Edwards and Glysson (1999) are presented by Wren et al. (2000) and Gray (2005). These alternatives can collect much more detailed data in a temporal sense and thus may produce more accurate data, but standardized methods and applicability to various conditions are not well established (Gray and Glysson, 2003; Gray, 2005).

DATA UNCERTAINTY

The issue of uncertainty in measured discharge and water quality data is often acknowledged but seldom addressed. Several recent developments, however, may cause the issue to receive increased attention in relation to water quality research, regulation, and modeling. First, TMDLs are required to include a margin of safety to account for uncertainty in load allocations (40 CFR 130.7). As TMDL litigation increases and intensifies, scientifically defensible uncertainty analysis will be needed instead of arbitrary estimates. In addition, the need for uncertainty estimates associated with model outputs has recently re-emerged because water quality models are increasingly used to guide natural resource decision-making (Beck, 1987; Hession et al., 1996; Sharpley et al., 2002). Because measured data uncertainty effects model output uncertainty, water quality modelers will be forced to consider input, calibration, and evaluation data uncertainty, potentially with the methods recommended by Moriasi et al. (2006). Finally, fundamental estimates of data uncertainty have recently become available to guide monitoring QA/QC efforts and watershed model evaluation (Harmel et al., 2006).

Uncertainty is introduced into measured water quality data by discharge measurement, sample collection, sample preservation/storage, and laboratory analysis (Harmel et al., 2006). The uncertainty associated with discharge measurement alternatives is well-established (Sauer and Meyer, 1992; Pelletier, 1988; Slade, 2004; Boning, 1992). For example, discharge measured in a properly designed weir (~10%) is more accurate than from a morphologically active natural channel (~20%) according to Slade (2004). Information on the uncertainty of sample storage, preservation, and analysis is also available (e.g., Lambert et al., 1992; Kotlash and Chessman, 1998; Ludtke et al., 2000; Jarvie et al., 2002). Similarly, cross-sectionally integrated sampling with EWI or EDI procedures is well understood and accepted as accurate (Wells et al., 1990; USGS, 1999).

In contrast, until recently, relatively little information on uncertainties associated with various automated water quality sampling procedures was available with which to select procedural alternatives. Sample collection procedures can, however, be the largest source of uncertainty in typical situations, according to Martin et al. (1992) and Harmel et al. (2006). Thus, sample collection procedures should receive substantial attention in design and implementation of all water quality monitoring projects and associated QA/QC plans. To accurately characterize water quality with available monitoring resources, typical monitoring projects with automated samplers should be designed to collect the type and number of samples to adequately capture constituent behavior (such as first flush and concentration hysteresis) without exceeding sampler capacity. Thus, the purpose of this article is to present information that assists project designers in achieving successful discharge and water quality data collection at the field and small watershed scale.

One limitation of a majority of the previous research on uncertainty related to water quality sampling is its focus on relative differences (precision) in error without regard to possible deviation from the true flux (accuracy). This limitation is attributed to the cost and commitment required to make true flux measurements. As a result, relative comparisons of various sampling strategies are available, but few attempt to quantify true uncertainty. A study by Harmel and King (2005), which was initiated to determine the uncertainty in measured storm water quality data from small agricultural watersheds, is one exception with regard to automated sampling. All fifteen of the flow-interval strategies evaluated (sampling intervals up to 5.28 mm volumetric depth with discrete sampling and composite sampling of 2 to

5 samples per bottle) produced cumulative load errors less than ±10%. The ranking of absolute errors in individual event and cumulative load estimation (sediment > NO_3-N > PO_4-P) is attributed to differences in within-event concentration variability as measured by the coefficient of variation (CV), which was also noted by Claridge (1975). The mean CV across sites for within-event concentrations was 0.61 for sediment, 0.39 for NO_3-N, and 0.19 for PO_4-P. The authors concluded that sampling intervals up to 6 mm should produce similar load accuracy in other locations for dissolved constituents such as PO_4-P that vary relatively little within runoff events, but smaller intervals (1 to 3 mm) should be used for sediment and particulate P.

Another exception is a study by Robertson and Roerish (1999), in which concentration data were collected "as frequently as economically possible." As a result, 90 to 195 samples were collected per site annually on eight watersheds (14 to 110 km^2). This study evaluated the ability of the ESTIMATOR regression method (Cohn et al., 1989) to determine annual loads from baseflow and storm water quality samples collected at various frequencies. Robertson and Roerish (1999) concluded that the regression method would at best produce median absolute errors of 30% for the small watersheds evaluated. In contrast, Haggard et al. (2003) indicated that the ESTIMATOR regression method was able to estimate annual constituent loads from a relatively large watershed within 10% of the loads determined with automated sampling.

SUMMARY

New and expanded efforts to characterize water quality on small watersheds are occurring across the U.S. This article describes state-of-the art data collection methods for discharge and water quality measurement. It is not meant to be an exhaustive guide but to describe appropriate methods along with advantages and disadvantages of alternatives. Extensive descriptions of the theory and procedures of discharge data collection are provided by fundamental sources such as Brakensiek et al. (1979), Buchanan and Somers (1976, 1982), Kennedy (1984), Carter and Davidian (1989), Chow et al. (1988), Haan et al. (1994), and Maidment (1993). Similar guides to water quality sampling are not as extensive because water quality is a more recent concern, but several sources provide general guidance on project design (Dissmeyer, 1994; USDA, 1996; USEPA, 1997). The USGS also provides extensive guidance on manual sampling (Wells et al., 1990; USGS, 1999). However, practical guidance on designing, implementing, and conducting automated storm sampling programs (e.g., McFarland and Hauck, 2001; Harmel et al., 2003; Haggard et al., 2003; Behrens et al., 2004) has only recently become available.

The amount of flow and water quality data becoming available to support water resource management is rapidly increasing. However, with recent initiation, expansion, and modification of projects designed to measure water quality, it is important to utilize data collection methods that accurately characterize discharge and water quality within the typical constraint of limited sampling resources. Based on current information on discharge and water quality sampling methodology, data collection activities should be conducted with the goal of providing high-quality (low uncertainty) data. The following general recommendations will increase the likelihood of achieving this goal:

- Consider wet-weather access, travel time, equipment costs, and sample collection method in the selection of site numbers and locations (Haan et al., 1994; USDA, 1996; USEPA, 1997).
- Commit sufficient personnel and financial resources to equipment repair and maintenance (USDA, 1996; USEPA, 1997).
- Assemble a well-trained, on-call field staff able to make frequent site visits (USEPA, 1997).
- Utilize reliable stage-discharge relationships, preferably accompanying pre-calibrated hydraulic control structures, for accurate discharge measurement (Brakensiek et al., 1979; Carter and Davidian, 1989; Haan et al., 1994; USDA, 1996; USEPA, 1997).
- Collect frequent, periodic manual grab samples to adequately characterize baseflow water quality (USDA, 1996).
- Collect flow-interval (or frequent time-interval) storm samples throughout the duration of runoff events to adequately characterize storm water quality (Richards and Holloway, 1987; Shih et al., 1994; Miller et al., 2000; Harmel et al., 2002; King and Harmel, 2003, 2004; Harmel and King, 2005).
- Use composite sampling to manage sample numbers without substantial increases in uncertainty (Miller et al., 2000; King and Harmel, 2003; Harmel and King, 2005).

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