3 Monitoring Plan Details

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In chapter 2 we discussed monitoring objectives, the fundamentals of good monitoring, the selection of an appropriate geographic scale for monitoring, and the selection of a basic monitoring design. In this chapter we discuss the nuts and bolts of monitoring, beginning with the selection of variables and concluding with data reporting and presentation. Because the emphasis of this guidance is placed on monitoring for watershed-level problem assessment, load estimation, trend analysis, and the effectiveness of BMPs or watershed projects, the details that follow in this and subsequent chapters will be centered on these objectives.

3.1 Variable Selection

Monitoring variables are often grouped into three general categories:

- Physical (e.g., flow, temperature, or suspended sediment)
- Chemical (e.g., DO, P, atrazine)
- Biological (e.g., *E. coli* bacteria, benthic macroinvertebrates, fish)

It is usually most appropriate for projects to monitor a mix of variables, although some projects may focus in one specific area such as physical measurements. Variables are often interrelated across these categories. For example, DO concentrations and temperature influence the fish assemblage present at a site.

Selection of the appropriate variables to monitor is a crucial task. A monitoring program cannot afford to measure every single variable nor should a project attempt to do so because some variables contribute to achieving project goals more than others. It is wasteful to measure characteristics that are unimportant or irrelevant to project objectives, and it is equally problematic to fail to measure key variables. In general, it is better to monitor a minimum set of variables well than a large number of variables poorly (e.g., minimal sampling frequency and/or duration).

3.1.1 General Considerations

The selection of which variables to measure in a monitoring program requires consideration of several important factors. It is important to resist the temptation to measure more variables than are needed for the project or to adopt a generic list of traditionally monitored water quality variables. The final design of a monitoring program often represents a compromise based on balancing information requirements, budget, personnel, and other constraints. Excess resources spent on analyzing unnecessary variables may force a reduction in the number of sampling stations, the sampling frequency, or the duration of monitoring, which can threaten program effectiveness.

The following sections discuss important factors to be considered when selecting variables to monitor. Variables commonly measured in watershed nonpoint source monitoring efforts are also discussed.

3.1.2 Selection Factors

Several factors should be considered when selecting variables to measure in a monitoring project. These factors, discussed below, are:

- Program objectives
- Waterbody use
- Water resource type
- Use impairment
- Pollutant sources
- Expected response to treatment
- Difficulty or cost of analysis
- Logistical constraints
- Need for covariates
- Priorities

3.1.2.1 Program Objectives

The overarching principle of monitoring variables selection is that the variables should be tied directly to the program objectives with due consideration of the other factors described in this section. In many cases, the stated program objective will clearly indicate the appropriate variable(s) to monitor. For example, an objective to document the effectiveness of BMPs on *E. coli* levels at a public beach clearly calls for measurement of *E. coli* bacteria. An objective to reduce TP loading to a lake would suggest measuring TP (perhaps not dissolved P) concentration and measuring flow because both concentration and flow data are required to calculate load (see section 3.8 and section 7.9). An objective to restore a fishery might require, at a minimum, monitoring the fish population as well as chemical (e.g., DO, ammonium) and physical (e.g., temperature, substrate) variables that support acceptable fish habitat.

It is more challenging to select monitoring variables when program objectives are less specific. For monitoring aimed at assessing water quality standards compliance or TMDL implementation, the selected variables should focus on what is required to assess water quality standards violations or TMDL achievement. For monitoring objectives that involve watershed reconnaissance or characterization, other factors such as the nature of the impairment, type of water resource, or likely pollutant sources must be considered.

3.1.2.2 Waterbody Use

Variable selection may be driven by a waterbody's designated use. Designated uses are one of three elements contained in water quality standards. The other elements are water quality criteria to protect those uses and determine if they are being attained, and antidegradation policies to help protect high quality water bodies (USEPA 2011c). States and tribes designate water bodies for specific uses based on their goals and expectations for their waters. Typical designated uses include:

- Protection and propagation of fish, shellfish, and wildlife
- Recreation
- Public water supply

- Agricultural, industrial, navigational, and other purposes.

Numeric and narrative water quality criteria are set to protect each designated use by describing the chemical, physical and biological conditions necessary for safe use of waters by humans and aquatic life. These criteria should be used to help guide variable selection and other monitoring details (e.g., sampling period and frequency) where use attainment or protection is the primary monitoring concern. Failure to meet some or all of the applicable water quality criteria can result in less than full support of designated uses.

For example, monitoring of a waterbody used for recreation might emphasize sediment, nutrient, or bacteria variables because these help define the aesthetic and health-related character of the waterbody. However, variables for monitoring irrigation water might include total dissolved solids and salinity variables and exclude less relevant biological variables. General applicability of water monitoring variable groups to selected designated uses is shown in Table 3-1.

3.1.2.3 Waterbody Use Impairment

Monitoring of waterbodies with documented use impairments can differ substantially from monitoring to assess use attainment or protection. For example, the impairment could be the result of a single pollutant (e.g., violation of a turbidity criterion) or failure to achieve one portion of a narrative criterion (e.g., fish assemblage), rather than a failure to meet all applicable criteria. In these situations, monitoring can be focused on the specific variables violating criteria instead of all potential variables indicated by the applicable water quality standard. While the variable list associated with criteria may be narrowed, additional variables should be considered to address the causes of the violation(s). For example, turbidity problems could be caused by streambank erosion, high phytoplankton production, or wash from impervious surfaces. Fish assemblage could be impacted by a number of factors such as lack of suitable flow or cover, water quality, or physical obstructions. For projects with an objective to relate water quality changes to pollution control efforts, it is essential to track variables associated with the causes of identified water quality problems.

3.1.2.4 Type of Water Resource Sampled

Variables monitored should be suitable for the type of waterbody under study. Appropriate variables often differ significantly between surface and ground water and between streams and lakes. Examples of variable groups that can be applicable to different water resource types are shown in Table 3-2.

3.1.2.5 Pollutant Sources

Variables monitored should reflect the nonpoint sources known or suspected to be present in the watershed. Crop agriculture, for example, is likely to influence suspended sediment, turbidity, nutrients and pesticides measured in water. The presence of intensive livestock agriculture in a watershed would justify measuring biochemical oxygen demand (BOD), nutrients and indicator bacteria. Urban stormwater sources are likely to influence variables such as discharge, temperature, turbidity, metals and indicator bacteria. Examples of variable groups that can be responsive to different nonpoint source activities are shown in Table 3-3.

Table 3-1. Monitoring variable groups by direct relationship to selected designated water use (adapted from USDA-NRCS 2003)

Variable	Designated Use				
	Aquatic life support	Contact recreation	Aesthetics	Irrigation	Drinking water supply
Physical					
Discharge	X				
Dissolved oxygen (DO)	X		X		X
Salinity	X			X	X
Secchi disk transparency	X	X	X		
Specific conductance	X			X	X
Suspended sediment	X	X	X		X
Temperature	X				
Total dissolved solids (TDS)	X			X	X
Turbidity	X	X	X		X
Chemical					
BOD	X		X		
Inorganics (Cl, F)	X			X	X
Metals (As, Cd, Cr, Cu, Fe, Hg, Pb, Zn)	X	X		X	X
Nutrients (N, P) – dissolved	X		X	X	
Nutrients (N, P) – total/particulate	X		X		
pH	X			X	X
Biological					
Benthic macroinvertebrates	X				
Chlorophyll a	X	X	X		X
Fish	X				
Indicator bacteria (fecal coliform, *E. coli*)		X			X
Macrophytes	X		X		
Pathogens (*Giardia, Cryptosporidium*)		X			X
Plankton (algae)	X		X		X

3.1.2.6 Response to Treatment

In a monitoring program designed to evaluate water quality response to management measure implementation, it is critical that monitored variables focus on dimensions of water quality expected to change in response to treatment. For example, an agricultural watershed uses conservation tillage as the principal management measure implemented to address an erosion problem. The water quality monitoring program should measure flow, peak flow, suspended sediment, and turbidity as variables likely to respond to widespread changes in tillage practices. It would be less appropriate to monitor for *E. coli*, even if *E. coli* standards are also violated in the watershed, unless land application of organic wastes in the watershed occurs in the watershed.

Table 3-2. Monitoring variables by selected water resource types (adapted from USDA-NRCS 2003)

Variable	Lake	Stream	Wetland	Ground Water
Discharge	X	X	X	
Dissolved oxygen	X	X	X	X
Habitat	X	X	X	
Riffle/pool ratio		X		
Salinity	X	X	X	X
Secchi disk transparency	X			
Specific conductance	X	X	X	X
Substrate characteristics	X	X	X	
Suspended sediment	X	X	X	
Temperature	X	X	X	
Total dissolved solids	X	X	X	X
Turbidity	X	X	X	
BOD	X	X	X	
Inorganics (Cl, F)		X	X	X
Metals (As, Cd, Cr, Cu, Fe, Hg, Pb, Zn)	X	X	X	X
Nutrients (N, P) – dissolved	X	X	X	X
Nutrients (N, P) – total/particulate	X	X	X	
pH	X	X	X	X
Benthic macroinvertebrates	X	X	X	
Chlorophyll a	X	X		
Fish	X	X	X	
Indicator bacteria (fecal coliform, *E. coli*)	X	X	X	X
Macrophytes	X	X	X	
Pathogens (*Giardia, Cryptosporidium*)	X	X	X	X
Plankton (algae)	X	X	X	

Research has shown that some BMPs can have unintended side effects. For example, increasing conservation tillage may result in increased herbicide use or increased concentrations and delivery of soluble nutrients. While conservation tillage has been shown to greatly reduce sediment bound P, P can become concentrated at the soil surface because of the lack of mixing by tillage, resulting in significant losses of soluble P in runoff (Beegle 1996). In these situations, it is advisable to monitor either TP or both particulate and dissolved P to ensure that BMP effectiveness is accurately assessed. Decisions on whether to track these variables, including adding subsurface monitoring sites, should be made at the beginning of a monitoring program.

Table 3-3. Monitoring variable groups by selected nonpoint source activities (adapted from USDA-NRCS 2003)

Variable	Nonpoint Source Activity				
	Crop Agriculture	Livestock Agriculture	Construction	Mining	Urban Stormwater
Physical					
Discharge	X	X	X	X	X
Dissolved oxygen	X	X		X	X
Salinity	X	X			
Secchi disk transparency	X	X	X		X
Specific conductance				X	X
Suspended sediment	X	X	X	X	X
Temperature			X	X	X
Total dissolved solids	X	X	X	X	X
Turbidity	X	X	X	X	X
Chemical					
BOD	X	X			X
Inorganics (Cl, F)				X	X
Metals (As, Cd, Cr, Cu, Fe, Hg, Pb, Zn)				X	X
Nutrients (N, P) – dissolved	X	X	X		X
Nutrients (N, P) – total/particulate	X	X	X		X
pH				X	
Biological					
Benthic macroinvertebrates	X	X	X	X	X
Chlorophyll a	X	X	X	X	X
Fish	X		X	X	X
Indicator bacteria (fecal coliform, *E. coli*)		X			X
Macrophytes	X	X		X	X
Pathogens (*Giardia*, *Cryptosporidium*)		X			X
Plankton (algae)	X	X		X	X

3.1.2.7 Difficulty or Cost of Analysis

The difficulty and cost of analysis must be considered in the selection of variables to monitor. While other factors like program objectives and pollutant sources should be more important criteria in the selection process, cost of analysis often drives choices among suitable variables because of budget constraints. Analytical costs will vary by region of the country and by laboratory. In-house laboratories, such as a university or a state agency, may have lower unit costs than an independent contract laboratory.

Some representative analytical costs are shown in Table 3-4. For several monitoring objectives, alternative monitoring variables that are lower cost may be available. For example turbidity analysis is half the cost of suspended sediment; a total dissolved solids measurement is about twice the cost of a laboratory analysis of specific conductance. Field measurement of conductivity is even cheaper if the equipment is available. These pairs of variables are likely to be highly correlated, making the lower cost alternative possibly the best choice (see section 3.1.3.3 for a discussion of surrogates). However, this will

not always be the case and cost alone should not be a primary criterion for variable selection. For example, a lower-cost analysis for NO_3-N ($17) measures an entirely different form of nitrogen from TKN.

Table 3-4. Representative laboratory analytical costs for selected water quality variables. Costs will vary by region and by laboratory (Dressing 2014)

Variable	Cost per analysis ($)
NO_3-N	17
TKN	35
TN	20
Soluble reactive P	15
Total P	22
Turbidity	8
Suspended sediment	16
Specific conductance	8
Total dissolved solids	15
Pesticide scan	135
COD	25
Oil and grease	45
Lead (ICP)	15
Invertebrates	150

It should also be noted that many variables can be analyzed by different methods that have both different costs and different levels of sensitivity. For example, a lead analysis by inductively coupled plasma (ICP) has a cost of $15/analysis using EPA method 200.9 (Barnstable County 2016) and a method detection limit of 0.7 µg/L (Creed et al. 1994). Compare this to a lead analysis by EPA method 200.5 with a method detection limit of 1.1 µg/L (Martin 2003) at a cost of $29/analysis (PSU 2016). Project objectives, data quality objectives and pollutant sources would factor into the trade-off between cost and sensitivity. Specific analytical methods can be further investigated in the *National Environmental Methods Index* (NEMI) at www.nemi.gov.

Finally, it should be noted that analytical costs, while potentially high, are often considerably lower than other categories of project costs, particularly personnel costs (see chapter 9). While cost alone is an important consideration, it cannot be the primary driver of variable selection. If monitoring of the appropriate variables cannot be correctly performed, money spent on monitoring is wasted.

3.1.2.8 Method Comparability

Advances in sampling and analytical methods are common. While these advances are welcomed on the one hand by reducing interference and improving reliability and accuracy, they can introduce challenges during the course of the project or when trying to design a new project that takes advantage of existing data. For example, it is wrong to compare historical turbidity data determined by the Jackson Candle method (units: Jackson Turbidity Unit or JTU) with turbidity data collected from a calibrated nephelometer (units: Nephelometric Turbidity Units or NTU). This caution extends to practically every phase of the monitoring program, from field sampling, sample preservation, and laboratory procedures. Ensuring that data from multiple methods can be compared is critical. One approach is to perform a comparability study by implementing both methods with laboratory splits and comparing the resulting

paired data. Depending on the results, it is prudent for projects with limited duration to continue with an older method rather than updating to a new method.

3.1.2.9 Logistical Constraints

Logistical issues like refrigeration availability at a sampling station or travel time between field sites and the laboratory may constrain selection of monitoring variables. Most water quality variables have specified permissible holding times and holding conditions. These parameters determine the length of time a sample can be stored after collection and prior to analysis without significantly affecting the analytical results. Maximum holding times and storage conditions have been established by the U.S. EPA (40 CFR 136.3, USEPA 2008b). Examples of these specifications are shown in Table 3-5.

Holding times and conditions will influence the choice of analytical variables. Unless samples can be delivered to the laboratory within six hours, *E. coli* analysis may be impractical. The demand for immediate filtration of samples for orthophosphate analysis may restrict that analysis to grab samples, while samples for TP can be held for 28 days. Samples for metals analysis can be held for up to six months before analysis, offering flexibility in analytical schedules and laboratory selection.

Table 3-5. EPA-recommended preservation conditions and hold times for selected water quality variables (40 CFR 136.3 and NEMI 2006)

Variable	Preservation	Maximum Holding Time From Sample Collection
pH	None	15 minutes
Ammonia	Cool, ≤6 ºC, H_2SO_4 to pH<2	28 days
Nitrate	Cool, ≤6 ºC	48 hours
Orthophosphate	Filter immediately, Cool, ≤6 ºC	48 hours
Total Phosphorus	Cool, ≤6 ºC, H_2SO_4 to pH<2	28 days
Total Dissolved Solids	Cool, ≤6 ºC	7 days
Specific Conductance	Cool, ≤6 ºC	28 days
Turbidity	Cool, ≤6 ºC	48 hours
Total Suspended Solids	Cool, ≤6 ºC	7 days
Pesticides	Amber glass bottle, sealed, Cool, 4 ºC	4 to 7 days depending on method
COD	Cool, 4 ºC, H_2SO_4 to pH<2	28 days
Oil and Grease	Cool, 4 ºC, H_2SO_4 to pH<2	28 days
Soluble metals (except Hg, B)	HNO3 to pH<2	6 months
E. coli	Cool, ≤10 ºC	6 hours

All of these constraints will drive station location, field schedules and staff requirements in a monitoring project. For example, in the St. Albans Bay Rural Clean Water project, samples from four tributary monitoring stations were analyzed for both orthophosphate and TP. This work required sample collection by a field technician two to three times each week in order to collect and retrieve samples and deliver them to the laboratory 28 mi (45 km) away (Vermont RCWP Coordinating Committee 1991). In contrast, the Lake Champlain Basin Agricultural Watersheds NNPSMP project collected weekly composite samples for P analysis that were analyzed for TP only, requiring a single trip by a field technician each week to retrieve samples and deliver them to the laboratory (Meals and Hopkins 2002). In both examples, power from the electrical grid was available to run the refrigerated samplers required to maintain sample temperatures at ≤6 ºC. Without power, there would be additional logistical challenges to keeping samples cold with ice or visiting stations more frequently.

3.1.2.10 Need for Covariates

It is important to consider monitoring variables not directly required by project objectives or pollutant sources but that may be important in understanding or explaining the behavior of other critical variables. Such explanatory variables that vary in concert with critical project variables are called covariates. Some covariates are obvious. For nonpoint source issues, precipitation and other weather variables are usually important covariates (see section 2.2.1). Even where load measurement is not required, flow (or stage) should always be measured, for example, as a key covariate in explaining observed patterns of suspended sediment or particulate P that are delivered predominantly in surface runoff in high-flow events. A monitoring program for a lake impaired by eutrophication may benefit from measurement of temperature, chlorophyll α, and algae, even if the focus is on reducing nutrient loads. In cases where paired watersheds are expected to have somewhat dissimilar hydrologic responses to precipitation events, it may be helpful to monitor additional variables such as instantaneous peak flow rate and average flow rate for inclusion in data analysis approaches (see section 7.8.2.2).

3.1.2.11 Set Priorities

Because numerous potential water quality variables exist and because selection criteria may conflict or overlap, it is useful to take a deliberate approach to setting priorities when designing a monitoring program. There are several ways to begin this approach. The USDA National Handbook of Water Quality Monitoring (USDA-NRCS 2003) recommends formulating a written justification for each candidate variable. If the justification is weak, the variable may be of low priority and might not be essential. A ranking system may be useful, where a minimum set of essential variables are identified (e.g., flow and TP for a TMDL aimed at a eutrophic lake), followed by a set of additional, justifiable variables to be monitored if other constraints allow (e.g., orthophosphate, nitrogen, Secchi disk transparency, and chlorophyll *a*). Finally, systematic evaluation of correlations among candidate variables may suggest that one variable (e.g., turbidity) is highly correlated to another (e.g., TSS) so that both need not be measured. Examination of such correlations may also show that some variables do not have direct covariates (e.g., NO_3-N) and should be given priority. Because some relationships between variables (e.g., turbidity and suspended sediment) can change as a result of watershed plan implementation (e.g., turbidity correlation with suspended sediment increases as nutrient levels and biological component of turbidity decrease), it may be appropriate to monitor both variables.

3.1.3 Physical and Chemical Water Quality Data

3.1.3.1 Measuring Surface Water Flow

Measuring surface water flow is an important component of many NPS water quality monitoring projects. Flooding, stream geomorphology, and aquatic life support are directly influenced by streamflow. Runoff and streamflow drive the generation, transport, and delivery of many NPS pollutants. Pollutant load calculations require knowledge of water flow (see section 3.8 and section 7.9).

Surface water *flow* is simply the continuous movement of water in runoff or open channels. This flow is often quantified as *discharge*, the rate of flow or the volume of water that passes through a channel cross section during a specific period of time. Discharge can be reported as total volume (e.g., acre-foot [ac/ft] or millions of gallons) or as a rate such as cubic feet per second (ft^3/s or cfs) or cubic meters per second (m^3/s). The depth of flowing water (m or ft) is commonly measured as *stage*, the elevation of the water surface relative to an arbitrary fixed point. Stage is itself important. Peak stage may exceed the capacity of stream channels, culverts, or other structures. Very low stage may stress aquatic life.

Flow data can be used for a variety of purposes, including problem assessment, watershed project planning, assessment of treatment needs, targeting source areas, design of management measures, and project evaluation. The selection of appropriate flow variables depends on the specific purpose and situation. Two common uses of flow data by watershed monitoring projects are pollutant load calculation (see section 7.9) and model calibration. Pollutant loads are critical elements of TMDL development and implementation. A pollutant load reduction is often one of the principal measures of success in NPS watershed projects. Discharge data are essential for the estimation of loads of sediment or chemical pollutants exported from a river or stream.

Evaluation of specific BMPs or watershed-scale BMP implementation often requires measurement of both pollutant concentration and flow. Many BMPs, particularly stormwater practices in urban settings, are designed to reduce total flow, peak flow, and/or flow velocity, as well as pollutant concentrations. The degree to which these practices achieve pollutant load reductions due to changes in flow versus changes in pollutant concentration varies. Careful consideration of the expected impacts of specific BMPs or combinations of BMPs should help guide decisions regarding flow variables to be monitored.

Basic principles of discharge measurement. Discharge is typically calculated as the product of *velocity* and *cross-sectional area* (Figure 3-1). Surface water *velocity* is the direction and speed with which the water is moving, measured in feet per second (ft/s) or meters per second (m/s). The cross-sectional area of an open channel is the area (ft^2 or m^2) of a slice in the water column made perpendicular to the flow direction.

Determination of discharge (usually symbolized as Q) thus requires two measurements: the cross-sectional area of the water in the channel (A, e.g., in m^2) and the area-weighted average velocity of moving water (V, e.g., in m/s). The product of these two measurements gives discharge in volume per unit time:

$$Q = V \times A$$

For example,

$$1.25\frac{m}{s} \times 36m^2 = 45\frac{m^3}{s}$$

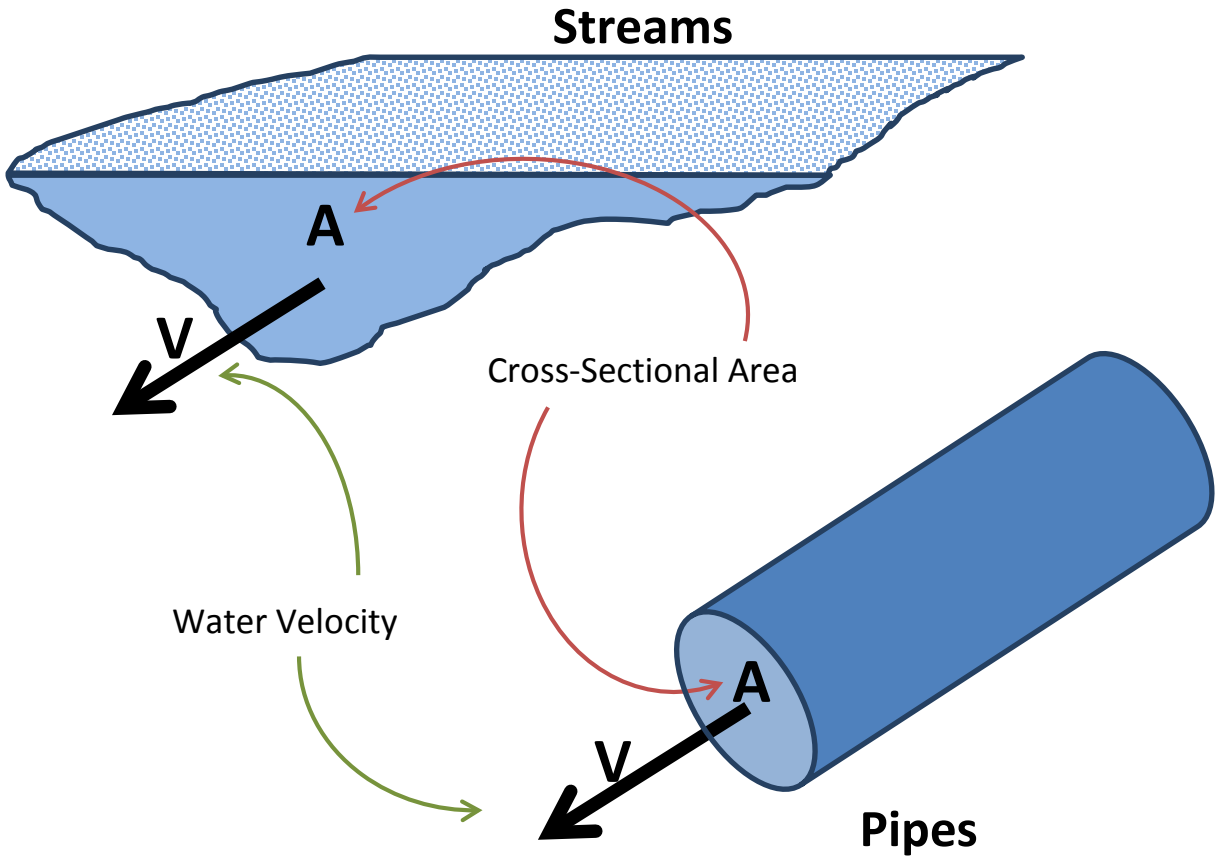


Figure 3-1. Cross-sectional area and water velocity for streams and pipes

It is important to recognize that the *velocity* of moving water varies both across a stream channel and from the surface to the bottom of the stream because of friction and irregularities in cross section and alignment – hence the use of average velocity in the above equation (see section 2.2.1.4.1). Friction caused by the rough channel surfaces slows the water near the bottom and sides of a channel so that the fastest water is usually near the center of the channel and near the surface. On a river bend, the water on the outside of the bend moves faster than the water on the inside of the bend, as it has to cover more distance in the same time frame. Clearly, more than a single measurement is needed to accurately characterize the velocity of water moving down the stream, particularly when the stream channel is irregular.

Flow measurement in water quality monitoring projects can take several forms, from a single measurement of peak stage during a high-flow event to continuous recording of stream discharge. Various approaches to measuring flow are described below.

Peak stage measurement. How high the water reaches during a storm event or flood, also known as peak stage, is often crucial information. In urban watershed projects where reduction of peak stormwater flows is a major goal, tracking peak stream stage (and precipitation) during storm events before and after watershed treatment can be a simple and inexpensive surrogate for monitoring actual streamflow. Peak

stage is important to determine for stream restoration projects where high flows shape the physical habitat of the stream. Peak stage is also essential to determine in flood planning, especially for flood frequency statistics, floodplain management, and design/protection of structures.

Peak stage can be observed by several informal means such as high water marks and debris lines on buildings or vegetation. More precise records of peak stage can be obtained using specialized crest gages (Figure 3-2). Information about crest gages is available at http://pubs.usgs.gov/fs/2005/3136/fs2005-3136-text.htm.

Figure 3-2. Traditional crest-stage gage

Instantaneous flow measurements. It is often necessary to estimate or measure discharge at a particular site at a particular time, either to document flow under certain conditions or to develop a data base for further analysis. There are several ways to determine instantaneous discharge, varying in accuracy and in applicability by the size of the stream.

- **Manning's Equation.** Discharge may be computed based on a slope-area method using the Manning equation:

$$Q = \left(\frac{1.486}{n}\right) A R^{\frac{2}{3}} S^{\frac{1}{2}}$$

Where:
Q = discharge in ft^3/s
A = mean area of the channel cross section in ft^2
R = mean hydraulic radius of the channel in ft
S = slope of the water surface in ft/ft
n = roughness factor depending on the character of the channel lining
1.486 = conversion factor in $ft^{1/3}/s$

The n factor can be estimated from tabular values and depends on the character of the channel, varying between 0.01 for smooth concrete to 0.10 for weedy streams with deep pools. The proper selection of a roughness factor is difficult in many cases and discharge determined by this method is only approximate.

- **Volumetric measurement.** For very small flows, e.g., low-flows in ditches or small streams or discharge from drain outlets, the most accurate method of discharge measurement is to simply

measure the time required to fill a container of known volume. In some circumstances, it may be necessary to use sandbags to temporarily channel flow to a practical collection point.

- **Dilution methods.** Dilution methods of discharge measurement consist of adding a concentrated tracer solution (salt or dye) of known strength to the stream and by chemical analysis determining its dilution after it has flowed far enough to mix completely with the stream and produce a uniform final concentration in the stream. Discharge is calculated as:

  $$Q = q * (C_1 - C_2)/(C_2 - C_o)$$

  Where:
  Q = stream discharge
  q = tracer injection rate
  C_1 = tracer concentration in injection
  C_2 = final concentration of tracer in the stream
  C_o = background tracer concentration in the stream

  The particular tracer selected should be conservative (i.e., slow to decay and not taken up by sediments or living organisms in the stream) and should be easily measured in the laboratory or field. Salt (NaCl) and rhodamine dye are commonly used tracers. Rhodamine dye can be analyzed in the field by fluorescence.

  When using tracers it is important to inventory all downstream uses of the water and check for notification requirements. Downstream users should be given advance notice of the study, including use of clear signage and other methods of communication.

- **Weirs and flumes.** For long-term projects, discharge can be measured using a weir or a flume, structures that water flows through or over that have a known relationship between stage and flow. If such a device is used, discharge measurement can be as simple as observing the stage of water just upstream of the device and consulting a table or using a simple equation to calculate discharge.

  Weirs are essentially dams built across an open channel over which water flows through a specially shaped opening or edge. Weirs are classified according to the shape of their opening – e.g., a 90º V-notch weir has a notch shaped like an inverted right triangle, whereas a rectangular weir has a rectangular notch. Figure 3-3 shows a 120º V-notch weir in operation. Each type of weir has an associated equation for determining the discharge rate, based on the depth (stage) of water in the pool formed upstream of the weir (see Rantz et al. 1982 for examples). In practice, weirs can range from small wood or metal plates temporarily mounted across small ditches or streams to more permanent installations involving concrete walls and other structures. Note that erecting any obstruction in a stream will create a pool upstream and care must be taken to avoid creating the potential for flooding during high flows.

  Flumes are specially shaped open channel flow sections that restrict the channel area, resulting in increased velocity and a change in water level as water flows through a flume. The discharge through a flume is determined by measuring the stage in the flume at a specific point, depending on the type of flume (see Rantz et al. 1982 for examples). In general, flumes are used to measure discharge where weirs are not feasible; flumes are often used to measure field runoff where flows during storm events can be collected and channeled through the device. Commonly used flumes include the Parshall (Figure 3-4) and Palmer-Bowlus (Figure 3-5). The H-flume is a special flume developed for agricultural field research that can measure discharge over a wide range with good

accuracy. Figure 3-6 shows an H-flume in operation in a field runoff monitoring project. Flumes come in a wide range of sizes denoting the maximum depth of flow they can accommodate and can be purchased as prefabricated units or built on-site. While flow control structures such as weirs and flumes can be pre-calibrated, the accuracy of discharge measurements can be compromised by faulty installation (Harmel et al. 2006, Komiskey et al. 2013).

Figure 3-3. 120º V-notch weir, Englesby Brook, Burlington, VT

Figure 3-4. Field application of small Parshall flume

Figure 3-5. Palmer-Bowlus flume

Figure 3-6. 2-foot (0.6 m) H-flume in place for edge-of-field monitoring, East Montpelier, VT

- **Area-velocity technique.** The most common method of measuring discharge in open channels is by measuring the cross-sectional area and the water velocity, as generally described earlier (Figure 3-7). Discharge in a small, wadeable stream can be measured by the following process:
  - **Select location.** Choose a straight reach, reasonably free of large rocks or obstructions, with a relatively flat streambed, away from the influence of abrupt changes in channel width.
  - **Establish cross-section.** Determine the width of the stream and string a cable or measuring tape across the stream at a right-angle to the flow. Divide the width into 20 to 25 segments (streams less than 10 ft [3 meters (m)]) wide may not allow as many segments) using tape or string to mark the center of each segment on the cable (Figure 3-8). Typically the stream is divided into enough segments so that each one has no more than 10 percent of the total streamflow.
  - **Measure depth of each segment.** At each mark across the stream, measure the depth from the water surface to the bottom with a graduated rod or stick (Figure 3-9).
  - **Measure water velocity.** At each mark, measure the velocity of the water (see below). Where depth is less than 2.5 feet (ft) (0.8 m), a single velocity measurement at 0.6 of the total depth below the water surface gives a reasonable estimate of the average velocity with respect to depth. For depths of 2.5 ft or more, the average of velocity measurements taken at 0.2 and 0.8 of depth is preferred.
  - **Calculate discharge for each segment.** For each segment, stream discharge is the product of width of the segment and the measured depth (giving area) multiplied by the velocity for that segment.
  - **Sum discharges.** Total stream discharge is the sum of all segment discharges.

Figure 3-7. Measuring stream discharge (USGS)

While wading is the preferred method for accurate discharge measurement, there are safety considerations that limit the flows at which wading can be accomplished. The USGS has a rule of thumb that prohibits wading if the product of depth (in ft) and velocity (in ft/s) exceeds 8 anywhere in the cross-section. Discharge measurement in larger rivers or at high flows follows the same principles of area and velocity but requires specialized techniques. These include suspension of equipment from bridges (Figure 3-10), cranes (Figure 3-11), or cableways, use of weighted sounding lines, and the use of heavy equipment for velocity measurement (Turnipseed and Sauer 2010).

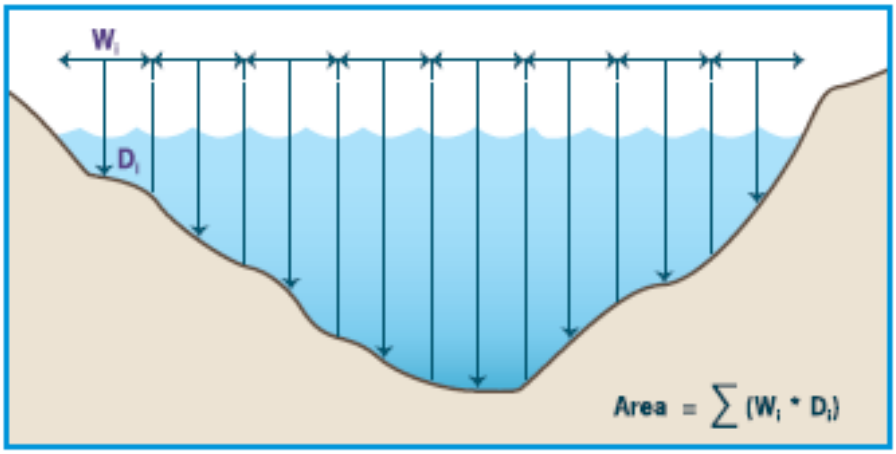


Figure 3-8. Delineation of stream-width segments for discharge measurement

Figure 3-9. Measuring the cross-section profile of a stream channel

Figure 3-10. Measuring discharge from bridge using an ADCP (acoustic Doppler current profiler) unit (USGS 2007)

Figure 3-11. Measuring discharge from a bridge using a current meter and crane (USGS n.d.)

Accurate velocity measurement is a critical component of the area-velocity technique. A variety of instruments are available to measure water velocity, from traditional mechanical current meters to electronic sensors (Turnipseed and Sauer 2010). Velocity measurement technology is evolving. For example, acoustic Doppler technology can measure velocity distributions within the flow, eliminating the need for wading or suspending instruments into the water (Fulton and Ostrowski 2008).

Continuous flow measurements. A single instantaneous measurement of stream discharge provides limited value because it provides information about only a single point in time. It is usually necessary to monitor discharge continuously when a project attempts to measure pollutant load over time or assess relationships between stream discharge and pollutant concentrations or aquatic life.

Continuous discharge measurement in open channels usually requires that the stage-discharge relationship is known, either through the installation of a weir or flume or through development of a stream rating. A stream rating is an equation determined for a specific site that relates discharge to stage based on a linear regression of a series of concurrent measurements of stage and discharge (e.g., by the area-velocity technique). Stage can be measured by a staff gage, a rigid metal plate graduated in meters or feet attached to a secure backing, linked through survey to a fixed elevation and located in a part of the stream where water is present even at low flows (Figure 3-12). Stage can also be read by measuring the distance from a fixed overhead point to the water surface (e.g., using a weighted wire or tape lowered from a bridge beam or using an ultrasonic sensor).

Figure 3-12. Staff gauge in stream

The rating equation should be based on measurements taken over a full range of streamflow conditions. It is usually unacceptable to extrapolate the rating equation beyond the range of observations that it is based on. As shown in the stream rating curve in Figure 3-13, stage-discharge relationships usually have a log-log form. With a valid stream rating, discharge can be determined simply from a stage observation plugged into the equation or read from a table. For more information on stage-discharge ratings, see http://training.usgs.gov/TEL/Nolan/SWProcedures/Index.html. Note that stream rating curves should be checked periodically, especially after major high-flow events. Rating curves frequently shift due to changes in streambed slope, channel roughness, and filling, scouring, or reshaping of streambanks.

Harmel et al. (2006) recommended against using Manning's equation in lieu of direct streamflow measurements to establish a stage-discharge relationship because it results in unacceptable uncertainty. In their analysis of various methods to estimate discharge they found that streamflow estimation with Manning's equation with a stage-discharge relationship for an unstable, mobile bed and a shifting channel resulted in a probable error range of ±42 percent. This compares with a range of 6 percent to 19 percent for typical scenarios using other methods.

Once the stream rating has been developed, continuous discharge measurement becomes an exercise in continuously measuring stream stage. Continuous measurement of stage is also used to record discharge through weirs or flumes where the rating is already known. Depending on the installation, this continual measurement can be accomplished in a number of ways.

A stilling well is a vertical tube or pipe hydraulically connected to the channel such that the level of water in the stilling well matches that in the channel, but the transient variations due to waves or turbulence are damped out (Figure 3-14). Stilling wells can range from an 8-inch-diameter (in) (20 centimeters [cm]) pipe connected to the side of a flume to a 3-ft-diameter (0.9 m) pipe placed in the ground and connected by pipes to a stream. Several devices exist to measure and record stage in a stilling well. Traditionally, this method was conducted using a float attached to a pulley that rose and fell with the water level in the well and moved a pen on a clock-drive chart recorder (Figure 3-15). There are modern versions that use electric chart drives or digital recording systems.

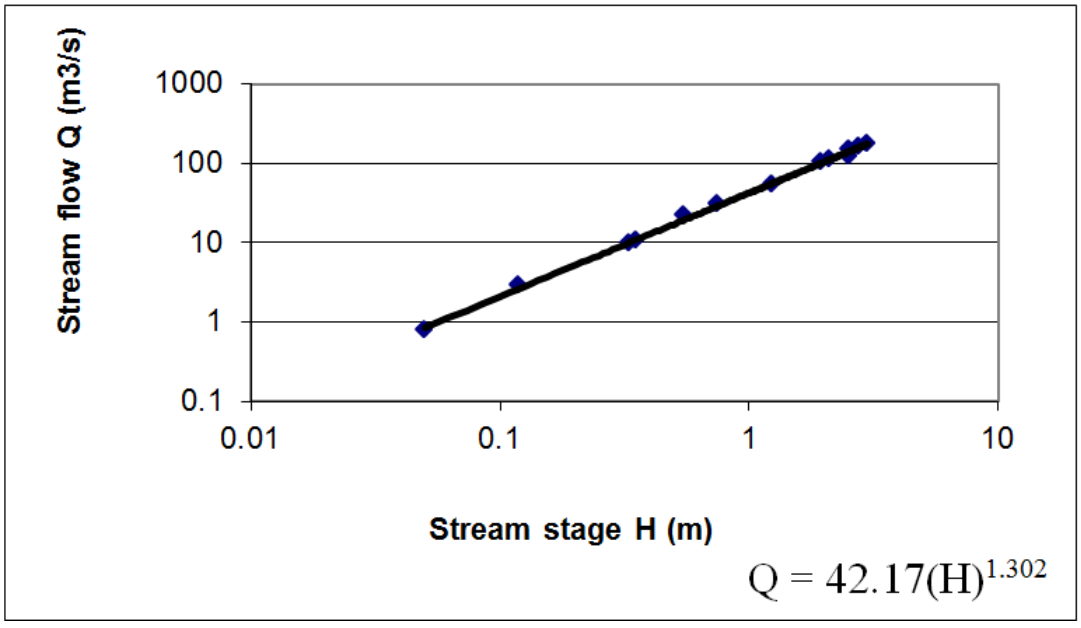


Figure 3-13. Example of a stage-discharge rating for a stream

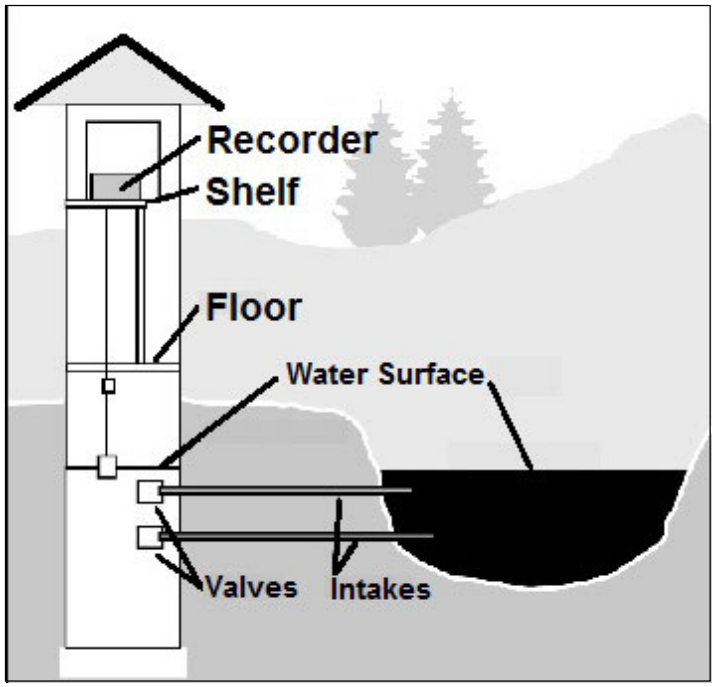


Figure 3-14. Stilling well design schematic (Wahl et al. 1995)

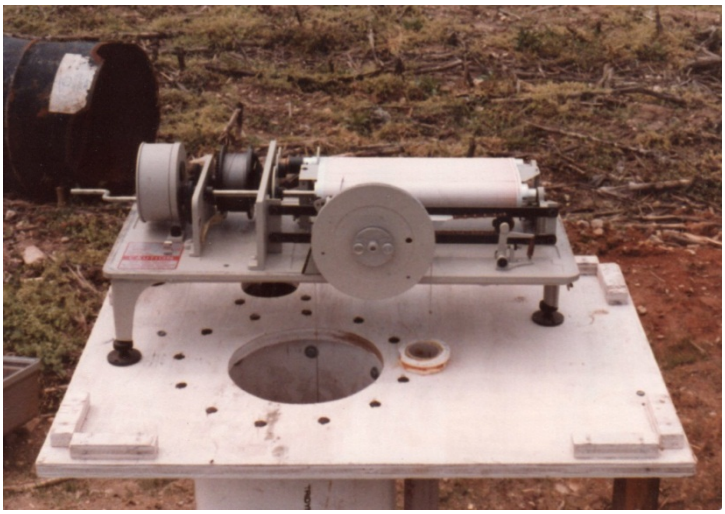


Figure 3-15. Traditional clock-drive chart recorder at a stilling well

Other approaches to measuring and recording level, either in stilling wells or directly in the channel include:

- **Bubblers.** Air or an inert gas is forced through a small diameter bubble line submerged in the flow channel. The water level is measured by determining the pressure needed to force air bubbles out of the line.
- **Pressure transducers.** A probe fixed to the bottom of the channel senses the pressure of the overlying water.
- **Ultrasonic sensors.** The sensor is mounted above the flow stream, and transmits a sound pulse that is reflected by the surface of the flow. The elapsed time between sending a pulse and receiving an echo determines the level in the channel.

Output from level recording sensors can either be recorded directly into a data logger (an electronic device connected to an instrument or sensor that records data over time) for later processing or into a specialized flow meter. There are several manufacturers of such meters; the meters often include the facility to calculate and record discharge and summary statistics, record other data such as precipitation, and interact with other devices such as automated water samplers.

Additional information on flow measurement can be obtained from:

- USDI Bureau of Land Reclamation. *Water Measurement Manual*. http://www.usbr.gov/tsc/techreferences/mands/wmm/WMM_3rd_2001.pdf.
- USGS Measurement and Computation of Streamflow. http://pubs.usgs.gov/wsp/wsp2175/

Streamflow measurement in a natural channel can be challenging for the novice, and mistakes made by technicians can greatly increase measurement uncertainty beyond the ranges reported by Harmel et al. (2006). This can result in highly unreliable stage-discharge relationships, inaccurate estimates of pollutant load, and spurious relationships between flow and other measured parameters. For these reasons, cost savings, and convenience, many monitoring teams seek flow data from USGS wherever possible. Real time daily stream flow data from USGS stations are available at http://waterdata.usgs.gov/usa/nwis/rt.

3.1.3.2 Commonly Measured Physical and Chemical Water Quality Constituents

Selected physical and chemical characteristics and constituents commonly measured in NPS monitoring programs are listed in Table 3-6. This is by no means an exhaustive list. These and other water quality variables are discussed in detail in the following sources:

- *Standard Methods for the Examination of Water and Wastewater* (Rice et al., 2012)
- U.S. EPA Clean Water Act Analytical Methods (http://water.epa.gov/scitech/methods/cwa/index.cfm)
- National Environmental Methods Index (www.nemi.gov/)

There are several complex issues associated with chemistry and analysis of some constituents that should be clarified. Below are a few brief discussions of some of the most important issues that those creating monitoring systems for NPS might encounter.

The traditional measurement of particulate matter suspended in water has been TSS, measured by filtering a subsample of water through a glass fiber filter and weighing the dried residue captured on the filter. In the last decade, research has reported a significant bias in the TSS analysis (Gray et al. 2000). The TSS analysis typically involves subsampling an aliquot from a bulk sample by pipette or pouring from an open container. This method often results in a significant underestimate of heavier particles (i.e., sand) in the sample and thus an underestimate of the total amount of suspended material in the original water. In contrast, the suspended sediment concentration (SSC) analysis entails measurement of the entire mass of sediment and the net weight for the entire sample, capturing all the particles in the original sample. An extensive comparative analysis (Gray et al. 2000) concluded that the TSS method frequently underestimates suspended sediment concentration and is fundamentally unreliable for the analysis of natural water samples. In contrast, the SSC method produces relatively reliable results for samples of natural water, regardless of the amount or percentage of sand-size material in the samples. SSC and TSS data collected from natural water are not comparable and should not be used interchangeably. NPS monitoring projects should monitor SSC, not TSS, to conduct accurate monitoring of suspended sediment loads. However, if comparability with past monitoring is required, it still may be necessary to measure TSS.

Table 3-6. Selected physical and chemical water quality variables commonly measured in NPS watershed monitoring programs

Variable	Abbreviation	Units	Definition	Notes
Physical Characteristics				
Salinity	-	g/kg mg/L	A measure of the total level of salts such as chlorides, sulfates, and bicarbonates in water.	Affects suitability of water (especially groundwater) for drinking, irrigation, and industrial use.
Secchi disk transparency	-	m	A measurement of water transparency in lakes using a black and white disc lowered into the water; the secchi depth is noted as the depth at which the pattern on the disk is no longer visible.	A common, inexpensive measurement of turbidity and an indicator of trophic status of lakes.
Specific conductance	COND	mS/m µmhos/cm	A measure of the ability of water to pass an electrical current; affected by the presence of inorganic dissolved solids.	Indirect measure of dissolved solids in water, highly correlated with salinity.
Total dissolved solids	TDS	mg/L	The sum of all dissolved matter (e.g., Ca, Cl, NO_3, P, Fe, S, and other ions) in a sample.	Indirect indicator of salinity; affects suitability of water for drinking, irrigation, industrial use.
Total suspended solids	TSS	mg/L	A measure of the weight of all particulate matter suspended in water obtained by separating particles from an aliquot of a water sample using filtration.	Affects water clarity, aquatic life support, suitability for drinking water and/or irrigation; indicates sediment from field and/or streambank erosion; particles carry other pollutants, e.g., P, metals, toxicants. It is a measure of wastewater treatment efficiency.
Suspended sediment concentration	SSC	mg/L	A measure of the weight of all suspended sediment in water obtained by separating particles from the entire water sample by filtration.	Related to TSS, but considered more representative of full range of particle sizes present in water because the entire sample, not a subsample, is filtered.
Temperature	T	° C	A measure of the thermal energy content of water.	Rates of biological and chemical processes depend on temperature. Solubility of oxygen is determined by temperature. Aquatic organisms from microbes to fish depend on certain temperature ranges for their optimal health, reproduction, and survival.
Turbidity	-	NTU	A measure of water clarity, i.e., how much suspended particulate material in water decreases the passage of light.	Indirect measure of suspended solids in water; particles may include soil particles, algae, plankton, microbes, and other substances.
Volatile suspended solids	VSS	mg/L	A measure of the organic portion of TSS.	Indicate what portion of TSS is organic in origin such as algal cells or organic wastes.
Nonvolatile suspended solids	NVSS	mg/L	A measure of the inorganic portion of TSS, usually calculated as the difference between TSS and VSS.	Indicate what portion of TSS is comprised of inorganic materials such as soil particles.
Chemical Characteristics				
Biochemical Oxygen Demand	BOD	mg/L	The amount of dissolved oxygen consumed by microorganisms in water in the decomposition of organic matter.	Indirect measure of organic pollutant levels. Usually referenced by oxygen consumed over specified time, e.g., 5-day BOD (BOD_5).
Dissolved Oxygen	DO	mg/L	Oxygen dissolved in water.	Supports aquatic life; influences form and availability of other pollutants.

Variable	Abbreviation	Units	Definition	Notes
Metals	(various)	mg/L or μg/L	Metals are trace elements having atomic weight from 60 – 200 (e.g., As, Cd, Cr, Cu, Hg, Ni, Pb, Zn). Metals exist in surface waters in colloidal, particulate, and dissolved phases; dissolved concentrations are generally low.	Behavior and toxicity varies by element, but metals generally exert chronic and/or acute health effects on aquatic organisms and humans. Presence of elevated concentrations may indicate influence of industrial waste, landfill leachate, or urban stormwater runoff.
Nitrogen – Ammonia	NH_3-N	mg/L	Unionized form of N produced by microbial mineralization of organic N.	Important plant nutrient; may contribute to eutrophication. Toxic to fish at high levels Results reported for ammonia N typically include both NH_3-N and NH_4^+-N forms.
Nitrogen – Ammonium	NH_4^+-N	mg/L	Ionized form of N produced by microbial mineralization of organic N.	Important plant nutrient. Under typical conditions, most ammonia in surface waters occurs as ammonium.
Nitrogen – Nitrite	NO_2-N	mg/L	A partially-oxidized form of N that is a short-lived product of mineralization and nitrification of N from organic materials, usually rapidly further oxidized to NO_3-N.	Nitrites have similar behavior and toxicity to nitrates; significant levels are rarely found in surface waters as they are rapidly converted to NO_3-N in aerobic environments.
Nitrogen – Nitrate	NO_3-N	mg/L	An oxidized form of N that is a common component of inorganic fertilizer; also produced by the mineralization and nitrification of N from organic materials.	Nitrates are highly soluble and mobile in surface and ground water; excess amounts can promote eutrophication and pose a health threat to humans and animals in drinking water.
Nitrogen – Nitrite + Nitrate	NO_2-N+NO_3-N	mg/L	Sum of nitrite and nitrate N in a sample.	Nitrite and nitrate are often analyzed together, depending on laboratory method
Nitrogen – Organic	-	mg/L	Nitrogen in a complex organic form (e.g., proteins) prior to mineralization to ammonia.	The presence of organic N indicates recent presence of organic wastes.
Nitrogen – Total Kjeldahl	TKN	mg/L	TKN is the sum of organic N and ammonia-N.	TKN includes all forms of N except NO_2-N and NO_3-N
Nitrogen - Total	TN	mg/L	Total N is the sum of all forms of N in a water sample.	TN can be determined directly through chemical analysis or calculated as the sum of TKN+ NO_2-N+NO_3-N
Phosphorus – Orthophosphate	OP PO_4-P	mg/L	The simplest and most stable of inorganic P compounds, H_3PO_4.	Ortho P (also referred to as “reactive” P) is a plant nutrient that may contribute to eutrophication.
Phosphorus – Soluble Reactive	SRP	mg/L	A dissolved form of P operationally defined as the P that reacts with specific reagents in a laboratory analysis.	Related and functionally similar to ortho phosphate, usually measured on a filtered sample.
Phosphorus – Total	TP	mg/L	Total P is the sum of all forms of P in a water sample, as determined by chemical digestion to a dissolved form.	Generally includes both particulate and dissolved P, unless operationally separated into dissolved and particulate forms in the laboratory.
pH	-	-	A measure of the acidity or basicity of water, expressed as the negative log of the H^+ ion concentration.	Affects chemical form of some pollutants, may have direct effects on aquatic life. Indicator of mine drainage.

Nitrogen (N) undergoes a complex cycle in the environment that includes both air and water pathways, mediated by microorganisms. A simplified N cycle is illustrated in Figure 3-16. Forms of N commonly measured as chemical water quality variables (Table 3-6) track the aqueous components of this cycle well. It should be noted that the term "ammonia" commonly refers to two chemical species that are in equilibrium in water (NH_3, un-ionized and NH_4^+, ionized). Water quality analyses for ammonia usually measure and report total ammonia (NH_3 plus NH_4^+). The toxicity of ammonia is primarily attributable to the un-ionized form (NH_3), as opposed to the ionized form (NH_4^+) (NCSU 2003). Ambient conditions of pH determine the net toxicity of total ammonia in water; in general, more un-ionized NH_3 and therefore greater toxicity exist at higher (alkaline) pH. NPS monitoring projects concerned with nitrogen should monitor total N, either as a discrete analysis or by measuring TKN and NO_2+NO_3 and summing the two for an estimate of total N unless there is a compelling reason to select different N variables.

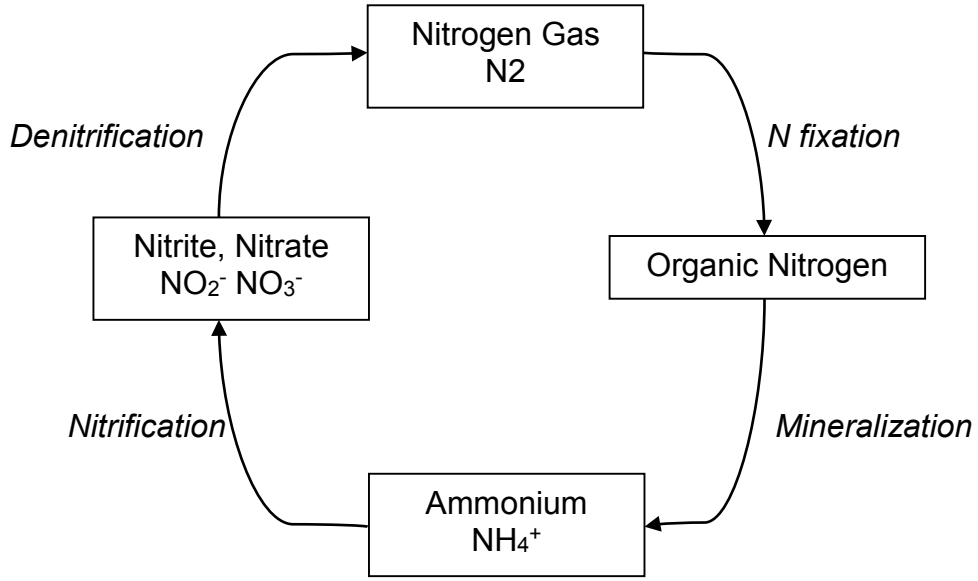


Figure 3-16. Simplified version of the nitrogen cycle

Phosphorus (P) undergoes a somewhat simpler cycle (USEPA 2012), lacking the atmospheric component, but the analytical scheme does not correspond perfectly to that cycle. In water quality monitoring, P is reported largely on an operational basis corresponding to sampling and laboratory procedures, rather than to specific points on a biogeochemical cycle.

P in freshwater systems exists in either a particulate phase or a dissolved phase. Particulate matter includes living and dead plankton, precipitates of P, P adsorbed to particulates, and amorphous P. The dissolved phase includes inorganic P (generally in the soluble orthophosphate form), organic P excreted by organisms, and macromolecular colloidal P. Most of these forms, however, are rarely analyzed specifically.

For some purposes, a water sample may be split between dissolved and particulate forms by filtration prior to further analysis. Thus, it is important to specify and know whether a specific P analysis is done for the dissolved phase, the particulate phase, or on the total sample. Ortho phosphorus is frequently analyzed as the primary dissolved form of P and is readily available to algae and aquatic plants. Most of the P discharged by wastewater treatment facilities is in the dissolved form. Another P fraction is also sometimes defined operationally as "reactive P" because it reacts with certain reagents in chemical analysis to form a color, resulting in reporting as Soluble Reactive P (SRP), which is related to but not

exactly equivalent to the ortho-P analysis[1]. To gauge the potential impact of a P discharge on eutrophication, "bioavailable P" is sometimes evaluated by measuring a sample's potential to support algae growth in a bioassay. Bioavailable P does not usually correspond exactly to a form of P directly measurable in chemical analysis.

Because the organic and inorganic particulate and soluble forms of P undergo continuous transformations (e.g., through uptake and release by algae and other plants or by chemical sorption and desorption on soils, suspended sediment, and other particulate material), many monitoring programs measure TP rather than individual forms to determine the amount of nutrient that can potentially support the growth of aquatic plants and contribute to eutrophication. The TP analysis uses digestion by acid and strong chemicals to convert all P in a sample to a soluble reactive form that can be easily measured in the laboratory (USEPA 2012). Several different digestion procedures are available and a monitoring program should be sure to specify the appropriate method for their situation.

3.1.3.3 Surrogates

In some cases, it may be preferable to use surrogate variables to represent other variables that may be mentioned specifically in project objectives but are difficult or expensive to measure. In some cases, it is necessary to use surrogates because a desired response variable is a complex composite of many individual factors. If, for example, the objective is to monitor the condition of salmon spawning areas, surrogate measures are necessary because the quality of spawning areas responds to many influences. Good surrogate variables would be stream bank undercut, substrate embeddedness, and vegetative overhang (Platts et al. 1983).

Two important criteria must be met by surrogate measures:

- A strong and consistent relationship must exist between the surrogate and the primary variable(s) of interest. Such a relationship can be established by simple linear regression using a local data set.
- A scientific basis is needed to assert that the surrogate and primary variable(s) will respond similarly to environmental management (e.g., BMP implementation) and change (i.e., the relationship remains the same). This assertion should be confirmed with data collected after such management or change.

While some surrogate relationships are widely appropriate in principle, the specifics of the relationship vary from site to site and, in most cases, should be based on locally derived data. For example, the relationship between turbidity and TP is usually highly specific to an individual watershed and should be used only in the system where the relationship can be documented. It is very important that the physical, chemical, or biological relationships between candidate surrogates and primary variables are considered in some depth to ensure that plausible relationships exist. For example, while erosion and sedimentation rates are often related in principle, using measured or estimated field erosion rates as a surrogate for watershed sediment load, for example, is likely to give poor results because the relationship between the two variables (i.e., the sediment delivery ratio) is not known.

[1] The term "orthophosphate" is a chemistry-based term that refers to the phosphate molecule all by itself (USEPA 2012). "Reactive phosphorus" is a corresponding method-based term that describes what you are actually measuring when you perform the test for orthophosphate. Because the lab procedure is not perfect, you get mostly orthophosphate but you also get a small fraction of some other forms.

Cost and ease of analysis are the primary reasons why specific conductance (fast and easy to measure with an electronic instrument) is often used as a surrogate for total dissolved solids (TDS) that requires measuring, drying and repeated weighing of a sample in a laboratory. In addition, specific conductance can be expected to respond to environmental management in the same way as TDS in many cases. Improved irrigation management, for example, might be expected to reduce levels of both actual TDS and specific conductance generated by the dissolved ions. Specific conductance can be expected to reflect the effect of irrigation management on TDS.

Indicator bacteria like *E. coli* are commonly used to indicate the likely presence of true pathogens in water because indicators are relatively fast and inexpensive to measure compared to pathogens. A good application of *E. coli* as a surrogate would be a study evaluating the effects of fencing livestock from streams because reductions in direct manure deposition to the stream would be expected to reduce both *E. coli* bacteria and manure-borne pathogens.

Turbidity is fast and easy to measure directly in the field and can be recorded continuously by field instruments. It is often highly correlated with TSS or SSC and can be used as a surrogate for these more expensive analyses when such correlations are established with local data. For example, if turbidity data will be used to predict or estimate TSS concentrations or loads (e.g., through a regression equation), the specific parameters of the equation must be documented in the local system because soils and suspended sediment vary widely among watersheds. In addition, turbidity can be a poor surrogate for SSC if the particles causing turbidity are not consistently related to those comprising SSC. Management changes that reduce SSC through trapping of larger sediment fractions, for example, may change the relationship between SSC and turbidity, which is more strongly linked to finer sediment. When turbidity is used as a surrogate for either TSS or SSC it is recommended that the relationship between the surrogate and primary variable is checked throughout the monitoring period to determine if changes have occurred.

3.1.4 Biological Data

Biological data, including aquatic organisms, habitat, and pathogens, are often central to NPS monitoring efforts. Selected biological characteristics commonly measured in NPS monitoring programs are listed in Table 3-7. This is not an exhaustive list. These and other water quality variables are discussed in detail in chapter 4 and in the following sources:

- Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish (Barbour et al. 1999).
- The Qualitative Habitat Evaluation Index (QHEI): Rationale, Methods, and Application (Rankin 1989).
- Methods for Assessing Habitat in Flowing Waters: Using the Qualitative Habitat Evaluation Index (QHEI) (MBI 2006).

Aquatic organisms are particularly useful because they integrate the exposure to various NPS pollutant stressors over time. Measures of biological communities can integrate the effects of different pollutant stressors like excess nutrients, toxic chemicals, increased temperature, and riparian degradation. They provide an aggregate measure of the impact of stressors from the watershed. When the objectives of a NPS watershed project focus on biological response (e.g., restoration of fish in a stream) or when treatment in the watershed focuses on in-stream practices like habitat restoration, biological monitoring is essential.

Table 3-7. Selected biological water quality variables commonly measured in NPS watershed monitoring programs

Variable	Units	Definition	Notes
Habitat variables			
Bottom substrate	Qualitative score	Percent rubble, gravel; presence of undercut banks, woody debris	Quality and diversity of substrate influences suitability for fish reproduction and habitat quality for benthic invertebrates.
Embeddedness	Qualitative score	Percent gravel, cobble, and boulder particles surrounded by fine sediment	Substrate condition influences suitability for fish reproduction and habitat quality for benthic invertebrates.
Flow velocity	cm/s	Range of current velocity	Prevailing current velocity influences suitability for stream biota.
Channel alteration	Qualitative score	Channelization, presence of point bars, silt deposition	Altered channels may reduce habitat diversity; sediment deposition can render substrate unsuitable for fish or invertebrate communities.
Pool/riffle ratio	Qualitative score	Variety of pool/riffle environments	A diversity or lack of pool and riffle environments influences suitability of a stream environment for fish and other biota.
Qualitative Habitat Evaluation Index (QHEI)[1]	Numerical score	Multiple metric index of habitat variables including substrate, cover, channel quality, riparian condition, bank erosion, pool/riffle distribution, drainage area, and gradient	The QHEI is composed of an array of metrics that describe attributes of physical habitat that may be important in explaining the presence, absence, and composition of fish communities in a stream. A significant correlation between QHEI and IBI has been documented in Ohio.
Microorganisms			
Indicator bacteria	#/100 ml cfu/100 ml MPN/100 ml	Bacteria of fecal origin whose presence is indicative of the probability of existence of true pathogens, e.g., fecal coliform, *E. coli*, *enterococci*	Use of indicator bacteria is based on rapid, inexpensive analysis, presumed association with true pathogens, and some epidemiological evidence of gastrointestinal disease.
Pathogens	#/L MPN/100 ml	Waterborne microorganisms that cause disease in humans or animals, including bacterial pathogens like *E. coli* O157:H7 and *Salmonella* and protozoans like *Giardia* and *Cryptosporidium*	Rarely analyzed as a routine because of expense and required expertise.
Microbial Source Tracking	-	Use of DNA, antibiotic resistance, or other techniques to attribute bacteria found in water to specific host group, e.g., human, cow, waterfowl	Increasingly used in situations of significant microbiological impairment where multiple sources are possible and specific cause(s) of impairment is unknown.
Plants			
Chlorophyll α	mg/L	Measurement of chlorophyll α pigment extracted from algae collected in a water sample	Used as an indicator of biological productivity or trophic state of lakes and as a surrogate for algal biomass. Often correlated with other measures of lake eutrophication such as P load and secchi disk transparency.
Algae	-	Identification and classification of algae taxa found in a sample of lake water	Presence and/or dominance of certain algal taxa may indicate trophic status of lakes, e.g., presence of *cyanobacteria* (blue-green algae) often indicate eutrophication due to excess P concentrations.
Macrophytes	(various)	Identification, classification, of macrophyte taxa and measurement of extent and abundance in a lake or stream	Presence of extensive growth of some species is considered a nuisance, especially invasive species; extent and abundance of other species is considered ecologically desirable.

Variable	Units	Definition	Notes
Benthic Macroinvertebrates			
# of organisms	#/m^2	Number of organisms found per unit area	Provides crude estimate of biomass for comparison between sites or over time.
Taxa richness	# of families	Number of families present	Reflects general health of community; generally increases with improving water quality, habitat diversity, and habitat suitability.
Biotic Index	Numerical score	Index based on tolerance of taxonomic groups (e.g., family) to organic pollution	Indicates general impacts of organic pollution on invertebrate community; values of the BI increase with decreasing water quality.
EPT Index	# of taxa	Number of distinct taxa within the *Ephemeroptera*, *Plecoptera*, and *Trichoptera* groups	Summarizes taxa richness within the insect groups that are generally considered to be pollution sensitive.
Functional feeding groups	(various)	Classification of organisms by feeding style, e.g., shredders, scrapers, filter-feeders, predators	Certain feeding groups indicate certain impairment types, e.g., shredders are sensitive to riparian zone impacts that change the inputs of coarse particulate organic matter to a stream.
Fish			
# of individuals	#/m^2	Number of individuals found per unit area	Reflects crude estimate of fish population size and biomass for comparison between sites or over time.
# of species	# of species	Number of different species present	Reflects general health of fish community; generally increases with improving water quality, habitat diversity, and habitat suitability. Presence/absence of particular species can be associated with water quality or particular stressors.
Index of Biotic Integrity	Numerical score	Integrated index of multiple metrics of species richness and composition, trophic composition, and fish abundance and condition	Individual component metrics can be used. IBI is adaptable and often modified on a regional basis.

[1] Developed by Ohio EPA (Rankin 1989)

Biological monitoring of macroinvertebrates, fish, and other aquatic biota must consider more dimensions than is the case for most physical and chemical monitoring. For example, the presence or absence of certain species or assemblages is not simply an indicator of ambient water quality or water quality impairment. The biotic community present at a particular location is always a reflection of the available habitat required to support those life forms. Data on aquatic biota cannot be interpreted without reference to the habitat at a particular site and is the reason that several habitat metrics are listed in Table 3-7. The nature of aquatic communities is also strongly determined by ecoregion. A warm-water river in Iowa is not capable of supporting the same biotic community as a Rocky Mountain stream in Montana. This is not necessarily because of a water quality impairment but because climate, watershed, soils, vegetation, and other factors differ between the two ecoregions. In NPS watershed monitoring projects, it is common practice to monitor biological variables at impaired or treated sites and at reference sites within the ecoregion indicating the best biological condition that can be expected in the subject watershed. For this reason, data on biological variables are often presented in comparison with data on the same variables collected at one or more reference sites. See chapter 4 for additional details on biological monitoring.

The use of indicator bacteria in biological monitoring is an evolving issue (Meals et al. 2013). Organisms like fecal coliform and *E. coli* are not themselves pathogenic but are assumed to have a significant association with the presence of true pathogens. Empirical evidence has suggested a statistical probability

of increased incidence of gastrointestinal disease at some threshold of indicator bacteria count (Dufour 1984). However, the adequacy of the association between indicator bacteria and true pathogenic microorganisms has been increasingly challenged in recent years (Harwood et al. 2005). Indicator organisms have been found in high numbers where few pathogens were detected and pathogens have been documented when a waterbody meets water quality standards for indicator bacteria. True pathogens like *Cryptosporidium* have been shown to survive considerably longer than *E. coli* in animal waste spread on agricultural land (Hutchison et al. 2005). Furthermore, the traditional presumption that indicator bacteria indicate recent fecal pollution is increasingly in doubt as fecal coliform and *E. coli* have been shown to survive for long periods and even reproduce in aquatic sediments, beach sands, and urban storm drains (Jiang et al. 2007, Yamahara 2009). However, other research continues to support an association between both *E. coli* and *enterococci* and the incidence of gastrointestinal disease (Arnone and Walling 2007), so the matter is far from resolved.

Indicator bacteria will likely continue to be widely used monitoring variables in the future. Water quality standards for shellfishing continue to be based on fecal coliform counts. TMDLs for bacteria are nearly always focused on fecal coliform, *E. coli* or some other indicator organism. As microbial source tracking becomes more widely cost-effective, that technology may become more important than simply measuring indicator bacteria counts at a sampling station (USEPA 2005a, 2011b). Furthermore, when waterborne disease outbreaks are an immediate concern, evaluation of true pathogens could be warranted.

See chapter 4 for a detailed discussion of biological monitoring approaches.

3.1.5 Weather Data

Weather is an essential variable set for NPS monitoring projects. Precipitation drives NPS pollutant generation and delivery and patterns of wet/dry weather, seasonality, and extremes are major influences on NPS loads. Actual weather data during a watershed project are needed to place the monitoring period in context with long-term average conditions. Weather is often a critical covariate in NPS projects, as unusually dry or wet weather may exaggerate or mask response to treatment. Precipitation variables like total rainfall, rainfall intensity, storm duration, and storm interval are often key design components in urban stormwater/LID practices. Temperature may be an important response variable in restoration of stream habitat and for implementation of urban stormwater BMPs. Finally, good weather data are usually key drivers for modeling and the extent and quality of precipitation data often determines the success of model calibration.

Variable selection is largely driven by specific project needs. In most cases, at least daily precipitation totals are needed. Data on storm intensity, duration, and frequency may also be needed where pollutant delivery is highly episodic and monitoring is focused on storm events. Air temperature (daily minimum, maximum, and mean) data may be needed because temperature drives evapotranspiration. In northern regions air temperature determines the form of precipitation as rain or snow and controls snowmelt. The majority of the annual NPS pollutant load in northern regions may be delivered by winter and spring snowmelt events (Hanson et al. 2000, Panuska et al. 2008). Other weather variables may be required by specific project objectives. For example, monitoring of stream fishery status after restoration of forested riparian buffers may benefit from data on solar radiation to correlate to shading and stream water temperature (Whitney 2007). A study of bacteria survival and transport in field runoff might need to monitor solar radiation, relative humidity, wind velocity, and soil temperature in addition to basic precipitation and air temperature as variables that affect microorganism survival after manure application.

Monitoring personnel should query local sources of weather data to determine the need for additional weather stations. A source of information for this step is:

- NOAA Earth System Research Laboratory. http://www.esrl.noaa.gov/psd/data/faq/
  - Provides information and links for locating climate and weather data and information.

Sources of current and historical weather data include:

- NOAA National Weather Service Internet Weather Source. http://weather.noaa.gov/
  - Provides weather conditions for the past 24 hours, forecasts, watches, and warnings. Data are easily copied and pasted into a spreadsheet.
- NOAA National Centers for Environmental Information Climate Data Online. https://www.ncdc.noaa.gov/cdo-web/
  - Provides for historical data retrievals and download to a comma delimited file.
- NOAA Meteorological Assimilation Data Ingest System (MADIS). https://madis.ncep.noaa.gov/index.shtml
  - MADIS is a meteorological observational database and data delivery system that provides observations that cover the globe. Data are available from July 2001 to the present.
- Weather Underground. http://www.wunderground.com/
  - Provides current and historical data that can be downloaded to a comma delimited file. Weather data come from more than 180,000 weather stations across the country.

3.1.6 Watershed Characterization

In designing any watershed monitoring program, it is essential to characterize the watershed to identify causes and sources of NPS pollution, understand how water and pollutants are transported through the watershed, and determine where and how to implement a monitoring program. In any specific project, data on particular watershed characteristics like geology or impervious cover may be needed, but in nearly all NPS projects, data on topography, soils, surface and subsurface drainage, hydrology (e.g., NHDPlus), and land use/land cover will be necessary. These data are often collected as part of the watershed project planning process described in detail by U.S. EPA (2008a).

3.1.6.1 Topographic Data

Topographic data may be needed to determine water flow paths, including mapping subcatchments, and to identify areas of steep slope, critical elevation, or particular aspect. Application of simulation models like SWAT (Soil and Water Assessment Tool) and AGNPS (AGricultural Non-Point Source Pollution Model) requires detailed topographic data. The main sources of topographic data in the recent past were published topographic maps. Today topographic data are readily available as Digital Elevation Models (DEMs) derived from remote sensing and assembled in a geographic information system (GIS). DEMs are commonly available from state or local agencies and, once imported into a GIS, can be readily manipulated to generate derived data on drainage area boundaries, hydrography, elevation, slope, and aspect.

A major consideration in DEM data for monitoring programs is resolution. Standard DEMs generally offer 30-meter resolution (i.e., vertical accuracy of ±30 meters), with 10-meter resolution possible in some cases, providing an improved representation of landscape features. Recent advances in using

LiDAR (*Light Detection and Ranging*, a remote sensing system using aircraft-mounted lasers) can provide DEMs with a resolution of 1 meter or better. High-resolution DEMs can be useful in locating and mapping very small-scale landscape features such as drainage ditches, swales, and ephemeral gullies, all of which can be important in understanding runoff and pollutant transport and identification of critical source areas to design land treatment.

USGS provides information on several sources of free geospatial data at:

http://education.usgs.gov/lessons/geospatialwebsites.html

3.1.6.2 Soil Characteristics

Data on soil physical characteristics and soil chemistry may be required for some NPS monitoring projects. Physical characteristics like hydrologic soil group strongly influence where surface runoff commonly occurs. Soil type and factors (e.g. soil erodibility) influence erosion and soil loss and are sometimes used as parameters to identify critical source areas of NPS pollutants in a watershed. Soil and vadose zone variables like permeability, hydrologic conductivity, or depth to water table may be important to determine in ground water monitoring efforts. Soil chemistry data (e.g., soil test P, organic matter, cation exchange capacity) may be essential to identify important source areas and understand pollutant transport over and through watershed soils. Testing for soil P levels can be helpful at the beginning of a project to ensure that paired watersheds, for example, are suitably matched (Bishop et al. 2005).

Data on soil characteristics may be available from specific studies in local areas or can be obtained from national databases such as the USDA *State Soil Geographic* (STATSGO) and *Soil Survey Geographic* (SSURGO) (http://soils.usda.gov/survey/geography/).

3.1.6.3 Land Use/Land Cover

Land use/land cover data includes information on the natural and cultural character of the land surface (e.g., forest, grassland, wetland, water, pavement) and on the activities taking place on the land (e.g., crop agriculture, pasture, residential, commercial, highways). Because NPS pollution is predominantly a function of land use, detailed knowledge of land uses and their spatial distribution is critical in developing a watershed monitoring program.

Land use/land cover data are usually derived from remote sensing data, either aerial photography or satellite imagery. Specific classification of land use/land cover types vary according to project objectives. For an urban stormwater/LID monitoring effort, data on many classes of developed land may be needed, as well as aggregate variables like impervious cover. In urban watersheds, the percent of direct and indirect impervious cover and other metrics of urban land use have been clearly documented as a determinant of many dimensions of stream impairment (Paul and Meyer 2001, Roy et al. 2003). In contrast, an agricultural NPS monitoring effort may need detailed information on many agricultural land uses like corn, soybeans, hay, pasture, farmstead but may lump urban land uses into a single broad category. A common land use/land cover classification scheme is shown in Table 3-8.

Table 3-8. Anderson Level II land use and land cover classification system for use with remote sensor data (Anderson et al. 1976)

1 Urban or Built-up Land	
	11 Residential
	12 Commercial and Services
	13 Industrial
	14 Transportation, Communications, and Utilities
	15 Industrial and Commercial Complexes
	16 Mixed Urban or Built-up Land
	17 Other Urban or Built-up Land
2 Agricultural Land	
	21 Cropland and Pasture
	22 Orchards, Groves, Vineyards, Nurseries, and Ornamental Horticultural Areas
	23 Confined Feeding Operations
3 Rangeland	
	31 Herbaceous Rangeland
	32 Shrub and Brush Rangeland
	33 Mixed Rangeland
4 Forest Land	
	41 Deciduous Forest Land
	42 Evergreen Forest Land
	43 Mixed Forest Land
5 Water	
	51 Streams and Canals
	52 Lakes
	53 Reservoirs
	54 Bays and Estuaries
6 Wetland	
	61 Forested Wetland
	62 Nonforested Wetland
7 Barren Land	
	71 Dry Salt Flats
	72 Beaches
	73 Sandy Areas other than Beaches
	74 Bare Exposed Rock
	75 Strip Mines, Quarries, and Gravel Pits
	76 Transitional Areas
	77 Mixed Barren Land
8 Tundra	
	81 Shrub and Brush Tundra
	82 Herbaceous Tundra
	83 Bare Ground Tundra
	84 Wet Tundra
	85 Mixed Tundra
9 Perennial Snow or Ice	
	91 Perennial Snowfields
	92 Glaciers

The land use/land cover variables of interest for watershed characterization are mainly static but are spatially referenced. A single map of current watershed land use/land cover may suffice for designing a water quality monitoring program; an annual update may be useful to relate to observed trends in water quality over time. Such data are distinct from land use activity data needed on a fine scale to relate to observed water quality at a site level, which include a critical temporal element. This kind of land use data monitoring is discussed later in section 3.7.

3.2 Sample Type Selection

3.2.1 General Considerations

The goal of collecting water samples is to obtain information representative of the target population for the monitoring effort. If monitoring is directed only at storm flows, the goal is to collect samples representative of storm flow conditions. If base flows are of greatest importance, then samples need to represent base flow conditions. For pollutant load estimation, it is most important that samples represent flow conditions that generate the greatest share of the pollutant load most strongly related to the identified problem. When monitoring is directed at specific conditions that threaten or harm aquatic life, sampling

may need to favor extreme conditions such as low flow or high temperature. Sample type choices can be a major determinant of the success or failure of a monitoring program.

As described in chapter 2, water quality varies both temporally and spatially. The extent that water quality spatial variability is addressed in a monitoring program is determined by the station location and the sample type. Station location determines where on the landscape a particular sample is taken, whereas sample type determines the spatial representation of each sample taken at that location. Similarly, sampling frequency and duration combine with sample type to determine the extent of temporal variability of water quality captured by the monitoring program. Sampling duration defines the timeframe for sampling, and sampling frequency determines how many times samples are collected during that timeframe. Sample type determines the degree to which temporal variability is captured within each sampling event.

There are generally four types of water quality samples (USDA-NRCS 2003):

- **Grab.** A discrete sample taken at a specific point and time.
- **Composite.** A series of grab samples collected at different times and mixed together.
  - Time-weighted: A fixed volume of sample collected at prescribed time intervals and then mixed together.
  - Flow-weighted: A series of samples each taken after a specified volume of flow has passed the monitoring station and then mixed together.
- **Integrated.** Multi-point sampling to account for spatial variations in water quality within a water body.
- **Continuous.** Truly continuous or very frequent sequential measurements using electrometric probes.

Each sample type has advantages and disadvantages and is discussed in the remainder of the next section. Ultimately, the selection of the appropriate sample type is determined by study objectives, variable(s) sampled, and whether concentration or mass is of interest (USDA-NRCS 2003). Integrated samples are generally preferred when suspended sediment is measured, and grab samples are preferred for bacteria. Generally appropriate sample type selection as a function of monitoring objective is illustrated in Table 3-9.

Table 3-9. Sample type as a function of monitoring objective (adapted from USDA-NRCS 2003)

	Sample Type				
		Composite			
Objective	Grab	Time-Weighted	Flow-Weighted	Integrated	Continuous
Problem Identification & Assessment	X	X	X	X	X
NPS Load Allocation			X		
Point Source Wasteload Allocation		X	X		
Trend Analysis	X	X	X	X	
Assess Watershed Project Effectiveness		X	X		
Assess BMP Effectiveness		X	X		
Assess Permit Compliance	X	X	X		

Objective	Sample Type				
	Grab	Composite		Integrated	Continuous
		Time-Weighted	Flow-Weighted		
Validate or Calibrate Models		X	X	X	
Conduct Research		X	X	X	X

3.2.2 Types

3.2.2.1 Grab

Grab samples are discrete samples taken from a specific point and time (USDA-NRCS 2003). For this reason, grab samples provide the narrowest representation of the spatial and temporal variability of water quality conditions. Grab samples are usually obtained manually with plastic or glass bottles or jars but can also be taken with automatic samplers. Grab sampling typically occurs in wadeable streams or from boats on lakes, but sampling can also be taken from bridges during high flows for larger streams and rivers. It is important to document both when and where grab samples are taken. Location can be recorded by recording depth and position along the width of the stream or depth and coordinates on a lake.

The specific method used to collect grab samples can have a significant influence on the content of the sample. Wilde et al. (2014) define samples for which the velocities of the stream and water entering the sampler intake are the same and different as isokinetic and nonisokinetic, respectively. Because the suspension of particulate materials depends largely on stream velocity, an isokinetic sample may therefore have a different and more accurate sediment concentration compared to a nonisokinetic sample. Isokinetic, depth-integrated samplers are described in section 3.2.2.3. Nonisokinetic samplers include the hand-held bottle, the weighted-bottle sampler, the BOD sampler, and the so-called "thief samplers" such as the Kemmerer and Van Dorn samplers that are often used for lake sampling at specific depths (Wilde et al. 2014).

3.2.2.2 Composite

Composite samples are generally considered a series of simple grab samples taken over time and lumped together (USDA-NRCS 2003). Isokinetic, depth-integrated samples collected to produce a discharge-weighted sample may also be included in this grouping (Wilde 2006). Composite samples are usually collected with automatic samplers (see section 3.6.2.4), but passive samplers (Bonilla et al. 2006) and labor-intensive manual methods can also be used. Composite samples derived from simple grab samples are taken from a single location and do not address the spatial variability of water quality conditions. When automatic samplers with fixed-depth intake(s) are used, the sample is considered by USGS to be a point-integrated sample (Wilde et al. 2014).

Sample preservation is always a concern but is of particular importance when automatic samplers are used for composite sampling. Analyte loss can occur between sample collection and laboratory analysis because of physical, chemical, and biological processes that result in chemical precipitation, adsorption, oxidation, reduction, ion exchange, degassing, or degradation (Wilde et al. 2009). Acidification and/or refrigeration is required for many monitoring variables.

The trigger for collecting samples distinguishes time-weighted from flow-weighted composite sampling. Time-weighted composite samples are derived from samples collected at pre-determined intervals such as

hourly or daily samples taken and composited in a single container (Stuntebeck et al. 2008). Because flow is not considered (but could be measured) in the sampling scheme, time-weighted composites are generally inappropriate for load estimation (see section 7.9) in nonpoint source applications. Where flow is constant, however, time-weighted composites would be useful for load estimation. If flow is measured in a time-weighted sampling scheme where samples are collected in multiple bottles, it is possible to make up a flow-weighted composite samples from individual discrete samples by adding amounts of individual samples in proportion to the flow that occurred over the collection interval (Stuntebeck et al, 2008). Peak pollutant concentrations may be missed in a time-weighted sampling design, however, resulting in low estimates of pollutant load.

Flow-weighted or flow-proportional samples are better for capturing the influence of both peak concentrations and peak flows, resulting in more accurate estimates of pollutant loads (see section 7.9). Collecting flow-weighted samples requires an established stage-discharge relationship, prediction of flow conditions during the period between sample collections, continuous flow measurement, and instantaneous and continuous calculation of flow volume that has passed the sampling station. Any fouling of the stage measurement by backflow, icing or other causes will result in incorrect flow volume calculations and the collection of non-representative samples. Remote access to the monitoring station provides some capability to address these potential problems. Flow-weighted composite sampling has as many applications as time-weighted composite sampling, with the additional advantage of being useful for pollutant load estimation (Table 3-9). The cost for flow-weighted sampling will exceed that of time-weighted sampling that does not include flow measurement. Both composite sampling types offer reduced laboratory costs per unit of temporal information gained when compared to grab sampling over the same time period because fewer samples are analyzed. Compositing results in information loss, however, as the individual samples are averaged either by time or flow. This information loss corresponds with reduced sample-to-sample variability which can be helpful in efforts to evaluate BMP and project effectiveness. For the same number of samples, composite sampling also offers the advantage of fewer trips to the field compared to grab sampling, reducing labor costs (see chapter 9 for a discussion of monitoring costs).

Advances in remote access and control of automatic sampling equipment have made it possible to adjust the sampling program based on current knowledge of weather conditions and discharge (Stuntebeck et al. 2008). This technology provides considerable flexibility for the researcher, including the ability to change flow-weighted sampling if flow conditions differ markedly from those assumed when the sampler was programmed.

3.2.2.3 Integrated

Grab samples can be integrated over depth and/or width. At flowing-water sites, USGS collects an isokinetic, depth-integrated, discharge-weighted sample as standard procedure (Wilde 2006). However, such a sample would not integrate temporal variations (USDA-NRCS 2003). Depth integration in lake sampling can be achieved by mixing grab samples taken from each lake stratum, by obtaining a simultaneous sample of the entire water column with a hose, or by automatic devices that collect at different depths over time (USDA-NRCS 2003).

Isokinetic, depth-integrating methods are designed to produce a discharge-weighted (velocity-weighted) sample (Wilde 2006). Using this method, each unit of stream discharge is equally represented in the sample, either by dividing the stream cross section into intervals of equal width (EWI) or equal discharge (EDI) (Wilde 2006). With the EWI method, depth integrated samples are collected at equally spaced intervals at the cross section and then composited (USDA-NRCS 2003). Under the EDI method, knowledge of stream discharge is used to divide the cross section into equal discharge subsections for

sampling. In theory, the two methods will produce composite samples with identical constituent concentrations. The instantaneous load could be determined by multiplying the analyte concentration by the measured instantaneous stream discharge (Wilde 2006). If nonisokinetic sampling methods are used, the method will not result in a discharge-weighted sample unless the stream is completely mixed laterally and vertically.

An isokinetic, depth-integrating sampler is designed to accumulate a representative water sample continuously and isokinetically from a vertical section of a stream while transiting the vertical at a uniform rate (Wilde et al. 2014). Isokinetic, depth-integrating samplers are categorized into either hand-held samplers or cable-and-reel samplers. The USGS provides details on how to use these devices for both isokinetic and nonisokinetic sampling (Wilde et al. 2014).

Integrated samples may be the best approach for situations where water quality is known to be spatially variable, e.g., vertical integration for lake sampling, or horizontal integration for river sampling. Given that the temporal variability of lake conditions is generally not as great as that in streams, integrated grab samples may be the most useful sample type for lakes. Grab samples at various lake depths, however, may provide necessary information that integrated samples "average out," so both types of samples could be appropriate depending upon the monitoring objectives. A combination of seasonal, integrated and simple grab samples taken at representative depths could be the best approach for problem assessment and trend analysis for lakes and other still water bodies. Composite or continuous sampling under these conditions would be likely to generate datasets with substantial serial correlation issues at a cost far greater than simple or integrated grab sampling.

The best approach for lake sampling will depend on project monitoring objectives and lake characteristics. Because sampling throughout the entire water column is not always necessary to characterize conditions of interest, integrated sampling can be unimportant. For example, when monitoring a vertically stratified lake for nutrient problems, it may be most desirable to collect surface grab samples for chlorophyll *a* and use meters to develop depth profiles of temperature, pH, conductivity, and DO. Nutrients could be monitored with surface grab samples only unless project objectives dictated that bottom samples were also necessary. Pairing the chlorophyll *a* and nutrient data from grab samples taken at various surface locations would be appropriate for analysis in most cases.

3.2.2.4 Continuous

Continuous sampling is not usually used in nonpoint source pollution studies, but the USGS uses continuous water-quality monitors in its national assessment of surface waters (Wagner et al. 2006). A commonly used configuration for USGS data collection is the four-parameter monitoring system, which collects temperature, specific conductance, dissolved oxygen, and pH data. Devices currently on the market have sensors for DO, conductivity, pH, turbidity, depth, chlorophyll α, blue-green algae, ammonia, NO_3, Cl^{-1}, total dissolved gas, temperature, and other parameters. Sondes are available that can measure 15 parameters simultaneously. Some instruments can store measurements to internal or external memory in a format compatible with a hand-held display, personal digital assistant (PDA), or laptop computer (Gibs et al. 2007).

Continuous sampling can be performed during short-term or long-term periods depending upon the monitoring objectives. Like grab and composite sampling, continuous monitoring provides no information about the spatial aspects of water quality conditions. Continuous sampling also has the potential to create information overload if carried out during a long period, with the potential consequence of expensive data reduction requirements, including addressing the problem of autocorrelation.

Other challenges associated with continuous sampling include the need for careful field observation, cleaning, and calibration of the sensors (Wagner et al. 2006). Despite manufacturer claims, even "self-cleaning" sensors require cleaning. Most electrodes are temperature dependent and many cannot be placed in areas of high stream velocity (USDA-NRCS 2003), but flow-through systems can be designed to address the stream velocity issue (Wagner et al. 2006). An advantage to continuous sampling is the ability to track the duration of values exceeding thresholds, in particular, those with significant diurnal variability.

With the exception of flow, continuous sampling is not frequently used in nonpoint source monitoring. It may be useful for variables such as temperature or dissolved oxygen, which should be measured *in situ* and for which minimum and maximum daily values are critical concerns. Continuous monitoring cost considerations include the cost of sondes and sensors, labor associated with keeping the sensors clean and operative, and costs associated with reducing the datasets for statistical analysis. Problem assessment and research are two areas for which continuous measurement could be highly appropriate. Continuous sampling could also track the exposure of aquatic organisms to harmful levels of temperature or DO, providing a very useful tool for trend analysis or an assessment of BMP or watershed project effectiveness.

3.3 Station Location

Monitoring station locations must be determined at two distinct scales. At the macro-scale, sampling locations must be determined by monitoring objectives, experimental design and resource type. The micro-scale issues of site access and physical configuration will drive the final selection of station locations.

3.3.1 Macro-scale

At the watershed or macro-scale, monitoring design (see section 2.4) will control station location. A single-watershed or trend design will require a station to be located at a watershed outlet where collected data represent water quality from the entire drainage area. An above-below or input-output design calls for two or more stations bracketing a treated area or an individual BMP to compare concentrations or loads entering and leaving the area. A synoptic or reconnaissance design will need numerous stations, located at areas that can isolate particular drainage areas or NPS pollutant source areas (Figure 3-17). Ground water monitoring for flow and/or mass determination will require an extensive network of monitoring wells to determine flow into and out of the area and to map hydrogeologic properties of the aquifer.

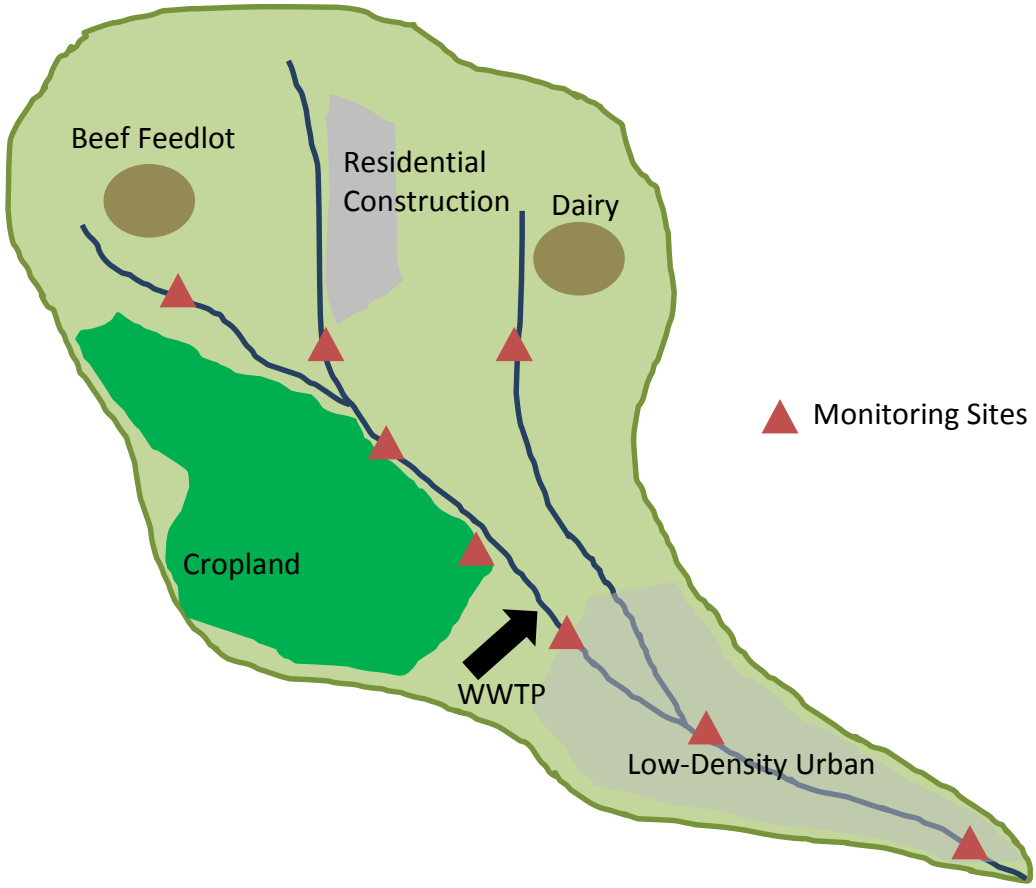


Figure 3-17. Possible sampling locations for a synoptic survey

Water body type is another macro-scale factor in station location. On stream or river networks, station locations might be selected to either capture or avoid the effects of tributary streams, to isolate sub-catchments, or to focus on areas of particular characteristics, e.g., high-quality regional biological reference sites. In lakes and reservoirs, monitoring stations at each major tributary discharge may be required to effectively measure load for a TMDL. In the lake itself, lake morphology, vertical stratification, and currents may require samples in several lake regions and/or at several depths in order to adequately represent water quality (Figure 3-18). Lake sampling designs and factors to consider when selecting sampling locations are described in detail by Nevers and Whitman (n.d.), and U.S. EPA (1998) provides guidance on sampling designs and locations for bioassessments.

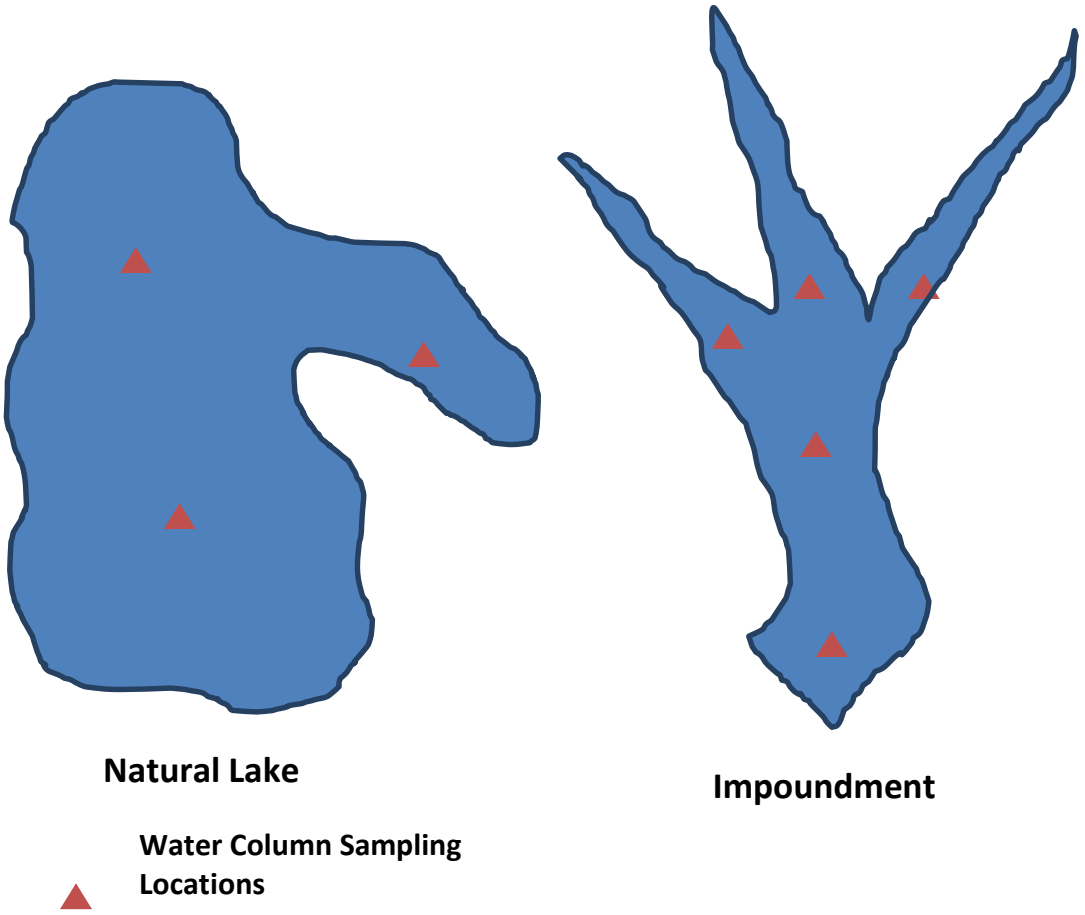


Figure 3-18. Potential lake monitoring locations

In its 2012 National Lakes Assessment, U.S. EPA randomly selected 904 natural lakes, ponds, and reservoirs across the lower 48 states using a probability based survey design (USEPA 2011a). To be included in the survey, these lakes (excluding the Great Lakes and the Great Salt Lake) had to be at least one meter deep and greater than 2.5 ac (1 ha) in size. In addition to these 904 sites, some sites were resampled for quality assurance purposes, and reference sites representing least-disturbed conditions were also sampled. A variety of field measurements were taken at "index sites" which are either the deepest point in a natural lake or the middle of a reservoir (USEPA 2011a). If the deepest point exceeded 50 m in depth, the index site was set as close to the middle of the lake as field staff could go without exceeding 50 m in depth. In addition, conditions of the littoral zone and shoreline were documented from stations around the lake.

The location of monitoring stations in ground water systems is determined by aquifer type and vertical, horizontal, and longitudinal variability in both water quality and water quantity (Figure 3-19). Both USDA-NRCS (2003) and Lapham et al. (1997) provide additional information on well selection for ground water monitoring.

In some cases, it may be appealing to adopt sampling stations that were part of a past monitoring network or are active in another project or program. Piggy-backing on past or existing monitoring stations may offer advantages of an historical data record or significant cost-savings. A prime example is co-locating with an operating USGS station. High-quality continuous flow data (sometimes in real-time through a

website) is a major benefit for a monitoring program because flow data are challenging and expensive to acquire. However, adopting sites from past or other monitoring programs must be carefully evaluated before decision making. Such stations may not be located optimally for the current monitoring program's objectives, and data collected for other purposes, objectives and schedules, or by other methods, may not be useful for current needs.

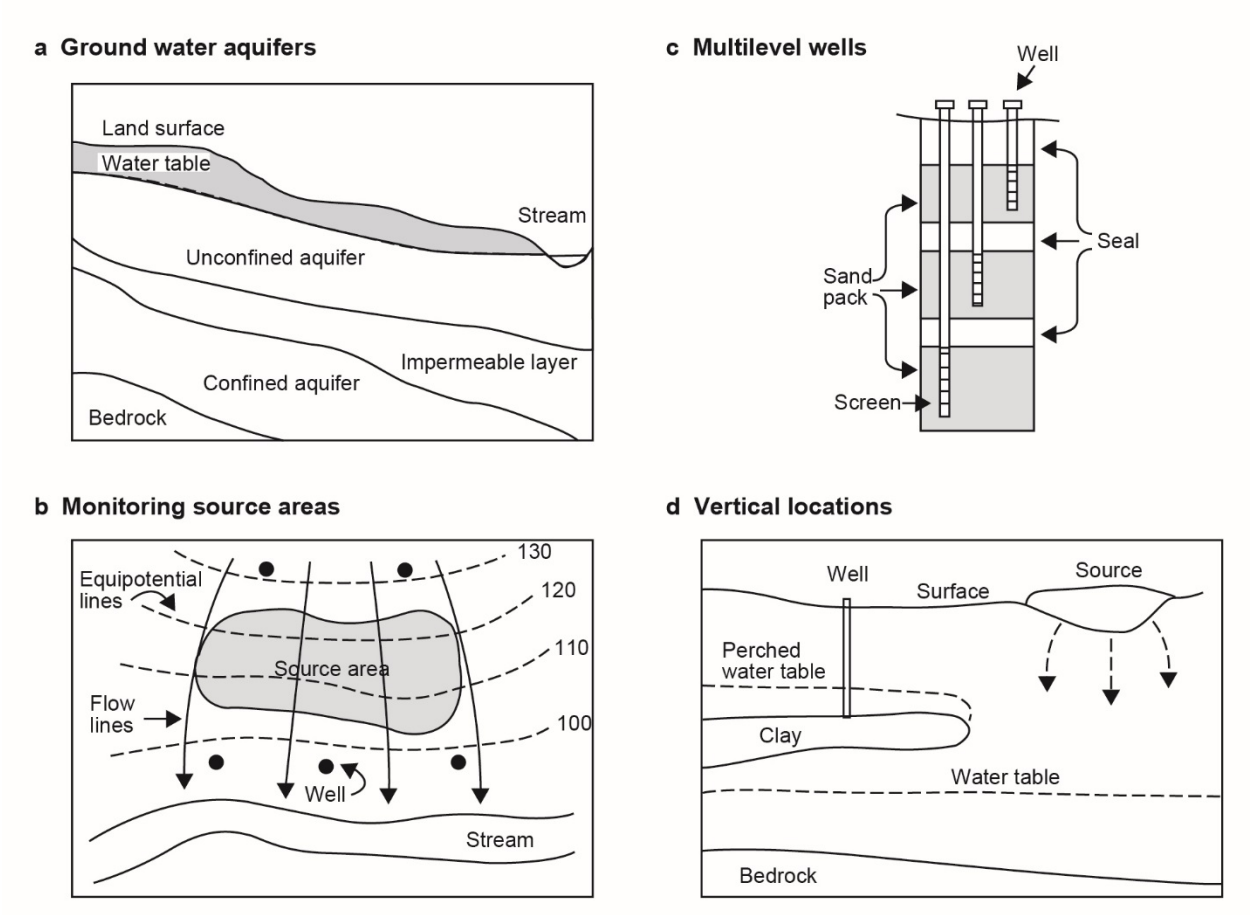


Figure 3-19. Possible groundwater monitoring locations (after USDA-NRCS 2003)

3.3.2 Micro-scale

Some general considerations apply to choosing the location of sampling stations at the local scale, and some specific factors apply to locations for flow measurement and biological monitoring.

3.3.2.1 General Considerations

Micro-Scale Site Location Considerations

- Representativeness
- Easy access
- Safety
- Power
- Permission
- Security

Stations must be located so samples and other data can be collected that are representative of the conditions being monitored according to project objectives. In practice, this means that stream stations should be located on relatively straight runs, away from obvious eddies or backwaters, far enough from major obstructions that prevent adequate mixing, and far enough downstream of tributary or other inputs to ensure complete mixing before samples are collected. Lake stations should be located far enough into open water to avoid obvious near-shore influences and outside of confined embayments unless near-shore or embayment conditions are of primary interest. In lakes of complex morphometry, multiple sampling stations may be required to collect representative data. Ground water sampling wells should be arrayed and installed in locations (both horizontally and vertically) that represent the resource of interest, e.g., a known contaminant plume or a regional aquifer system.

Many relevant general considerations for local-scale station location relate to practical matters of logistics (see section 2.2.3.1). Access, in terms of both travel from a base to the site and foot access to the stream and/or station facilities, is critical. Considering the safety of field staff, especially in harsh seasons or inclement weather, is vital. The availability of power and communication links may be essential to some station types. Security, from both human interference and natural threats like flooding, is important as is land ownership. In some cases, stations can be located in the highway right-of-way or on a bridge structure, avoiding the need for negotiations with private landowners (although permission/approval from the state or local transportation agency is usually required). In some cases, permission from or lease or rental agreements with property owners may be required. Finally, if buildings, electrical power, or other physical structures are to be installed, local land use permits may be required.

3.3.2.2 Locations for Flow Measurement

There are some special considerations for locating stream stations at which flow will be measured in open channels.

- Select a straight reach, reasonably free of large rocks or obstructions, with a relatively flat streambed, away from the influence of abrupt changes in channel width.
- Avoid culverts, waterfalls, and bridges where obstructions or degraded structures may cause hydraulic anomalies that interfere with a stable stage-discharge relationship.
- Seek an area with a stable cross-section and avoid areas subject to frequent deposition of sand or gravel bars or severe bank erosion.
- Look for an area where depth and velocity measurements can be conducted safely at low flows.
- Look for an area where a bridge crossing or walkway allows safe velocity measurements at high flows.
- Look for areas where stage can be measured and/or recorded continuously, e.g., protected area for a staff gage.

Where flow is to be measured at the edge of a field or elsewhere using a weir or a flume, look for sites where flow can be collected and/or diverted into the device, where ponding caused by a weir will not

cause problems, and where concentrated discharge from a flume can be safely conveyed away downstream. See section 3.1.3.1 for additional information on flow measurement.

3.3.2.3 Locations for Biological Monitoring

Rapid Bioassessment Protocols (Barbour et al. 1999) lists several important considerations for locating biomonitoring sites.

- Ensure a generally comparable habitat at each station. Otherwise, differences in biology attributable to local habitat alone will be difficult to separate from differences or changes in response to water quality degradation due to NPS pollution.
- Locally modified sites, such as small impoundments and bridge areas, should be avoided unless project objectives are to assess their effects.
- Sampling near the mouths of tributaries entering large waterbodies should be avoided because these areas will have habitat more typical of the larger waterbody.
- Biological monitoring programs generally require a reference site to provide data on the best attainable biological conditions in a local or regional system of comparable habitat.

See chapter 4 for additional information on locating biological monitoring sites.

3.4 Sampling Frequency and Duration

The questions of how often to collect samples (the sampling frequency or interval between samples) and how long to conduct a sampling program are critical and without simple or stock answers. The choice of sampling frequency depends on program objectives, type of water body involved, variables measured, and available budget.

3.4.1 General Considerations

In general, sampling frequency must be relatively high (e.g., daily to weekly) for monitoring to evaluate effectiveness of a single BMP or to document the mechanisms controlling water quality at a particular site. Automatic samplers with flow meters that can collect composite, flow-weighted, samples over storm events and collect weekly or biweekly samples enable effective sampling for concentration and load data to evaluate BMP effectiveness. They also reduce the high cost of retrieving and analyzing samples collected more frequently. A program with an objective of detecting a long-term trend or evaluating watershed program effectiveness can accept longer intervals (e.g., weekly to monthly) between samples. Considerations specific to monitoring for load estimation are discussed in section 3.8.

Sampling frequency must also be determined based on the type of waterbody being monitored, and in particular the variability of water quality in the waterbody. Greater variability requires higher sampling frequency to obtain a reasonable picture of water quality. For example, water quality in edge-of-field runoff from cropland is likely to be considerably more variable and require considerably more frequent sampling than water quality in a large lake or a regional aquifer. Water quality in intermittent streams is usually more variable than in large river systems. A general guide to the relationship between system variability and sampling interval is illustrated in Figure 3-20.

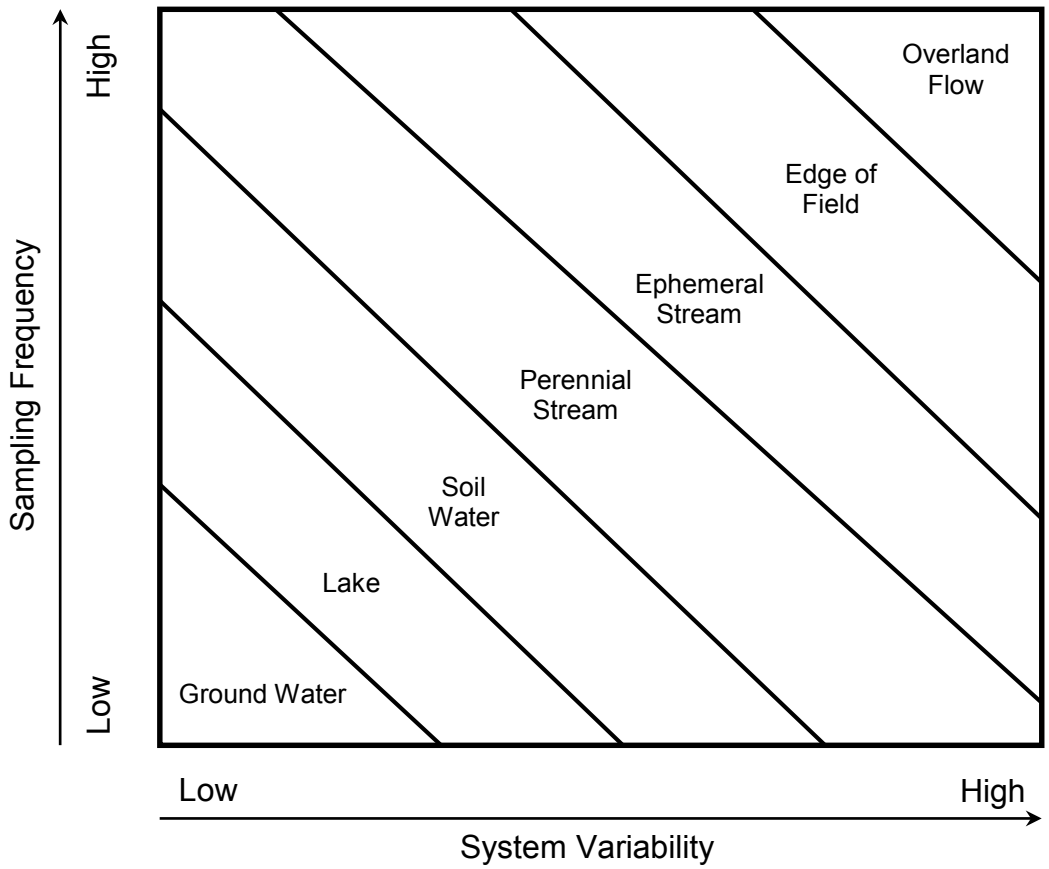


Figure 3-20. Schematic of sampling frequency as a function of system type (after USDA-NRCS 2003)

Project budgets, staff availability, and laboratory capability typically put limitations on sampling frequency, but financial resources should not be the primary basis for decisions on sampling frequency. A sampling program that cannot achieve desired objectives because of inadequate sampling frequency is not cost-effective. Where resources are limited, consider reducing the list of variables analyzed or even the number of stations before cutting back on the sampling frequency that is required to meet project objectives. Use of less expensive surrogate variables, simplifying field instrumentation, and the use of composite sampling programs are all ways to reduce costs while maintaining the critical sampling frequency.

Calculation of appropriate sampling frequency varies with the statistical objective for the monitoring data and sampling regime. Following are examples of how sampling frequency in the context of simple random sampling can be calculated for estimating the mean and for detecting trends.

3.4.1.1 Estimating the Mean

A common objective for monitoring data is to be able to estimate the mean value of a water quality variable with a specific level of confidence in the estimate. The equation for calculating the sample size (Reckhow and Chapra 1983, USDA-NRCS 2003) is:

$$n = \frac{t^2 s^2}{d^2}$$

where:
n = the calculated sample size
t = Student's t at n-1 degrees of freedom and a specified confidence level
s = estimate of the sample standard deviation
d = acceptable difference of the estimate from the estimate of the true mean, or ½ of the confidence interval from the mean

The t value is taken from a table of Student's t at the desired confidence level (*P*) (typically 0.05 or 0.10). In general, a two-tailed t-test should be used because we are usually interested in error on both sides of the mean. The estimate of the population standard deviation is best obtained from baseline data from the monitored water body; if such data are lacking, an estimate from a comparable nearby system can be used. The acceptable confidence interval from the true mean can be expressed as a percent of the mean. The actual calculation may be an iterative process because the value of t may change with the particular value of n chosen. See file nmean.pdf for an example.

3.4.1.2 Detecting a Step or Linear Trend

Another objective for monitoring data might be to detect a change or trend in the value of a water quality variable with a specific level of confidence (see section 7.8.2.4 for a discussion of trend analysis techniques).

Commonly in watershed studies, there are two types of change in the water quality variable studied:

- a step change that compares the pre- and post- water quality mean values
- a linear (gradual, consistent) trend over time

To determine sample size to detect a step change (e.g., comparing the change in baseline mean due to implementation of land management changes), the detectable change must first be calculated based upon the standard deviation of the difference between the pre- and post- means with an anticipated number of samples. See section 3.4.2.3 for an example calculation to determine the detectable step change with a given sample size. With an iterative process of trying different pre- and post- sample sizes, a sample size to detect a step change difference of acceptable magnitude can be estimated. See file ntrend.pdf for an example.

As with documenting a step change between pre- and post- BMP periods, monitoring for trend detection must be sensitive enough to detect the level of water quality change likely to occur in response to management changes. For a linear trend, this monitoring is based upon the confidence interval on the standard deviation of the slope. The standard deviation of the slope (S_{b1}) is a function of both the square root of the MSE (which is the standard deviation of the water quality data with any linear trend removed), as well as the spread of the X's (in this case, length of monitoring:

$$S_{b1} = \frac{\sqrt{\text{MSE}}}{\sqrt{\sum(\text{Xi} - \overline{\text{X}})^2}}$$

Where: MSE = standard deviation of the water quality data with any linear trend removed
X_i = X value at time i, $\overline{X}$ is the average X values.

Typically, for watershed studies, X is expressed as a 'DATE' value which represents 1 day. The slope is therefore expressed as change per day. To express as a change per year over N number of years, the slope/per day would be multiplied by 365 days/year and N number of years.

Therefore, for a linear regression of water quality values vs. time, one-half of the confidence interval on the slope is:

½ confidence interval = (N) * $t_{(n*N-2)df}$ * 365 * s_{b1} [same as Minimal Detectable Change]

Where: $t_{(n*N-2)df}$ = One-sided Student's *t*-statistic (α=.05)
N = Number of monitoring years
n = Number of samples per year
df = degrees of freedom
365 = Correction factor to put the slope on an annual basis when DATE is entered as a Date (day) variable, e.g., the slope is in units per day. If DATE values were 1-12 for months and the slope was expressed 'per month' then this value would be "12."

The sample size could therefore be calculated interactively by trying various sample frequencies and durations until the watershed monitoring would be able to detect the amount of change anticipated by BMP implementation.

If pre-BMP data exist, the sample variance can be used to estimate MSE (or capture the MSE by running the sampled data through a linear regression computer program. Table 3-10 gives sample size for common sample intervals and durations. Table 3-11 provides example values of $\sum(Xi - \overline{X})^2$ for biweekly sampling that were generated using P concentration data from a long-term NPS monitoring project. This information is used in the linear trend example in file ntrend.pdf. Note that the required sample duration will increase when corrected for autocorrelation (See section 3.4.2).

See Spooner et al. (2011) for more details on calculating the minimum detectable change (MDC) for linear trends. See file ntrend.pdf for an example (hyperlink to be added).

Table 3-10. Number of total samples per indicated sample frequency and number of years

	Total number of samples, n		
Number of years, N	Weekly	Biweekly	Monthly
1	52	26	12
2	104	52	24
3	156	78	36
4	208	104	48
5	260	130	60
6	312	156	72
7	364	182	84
8	416	208	96
9	168	234	108
10	520	260	120

Table 3-11. Values of $\sqrt{\sum(Xi - \bar{X})^2}$ for biweekly sampling for selected monitoring durations, assuming X_i is measured as a 'Date' or daily variable

Number of years, N	$\sqrt{\sum(Xi - \bar{X})^2}$
2	1,472
4	4,224
8	15,955

3.4.2 Minimum Detectable Change (MDC) Analysis

3.4.2.1 Definition and Overview

The MDC is the minimum change in a pollutant concentration (or load) during a given time period required for the change to be considered statistically significant. Most of the material presented is taken from Spooner et al. (2011) where the reader will find a more detailed discussion, relevant equations, and illustrative examples.

The calculation of MDC has several practical uses, including determining appropriate sampling frequencies (discussed here) and assessing whether a BMP implementation plan will be sufficient for creating change that is measurable with the planned monitoring design (see section 7.6.3). The same basic equations are used for both applications with the specific equations depending primarily on whether a gradual (linear) or step trend is anticipated. The reader is referred to Meals et al. (2011b) for a discussion of these types of trends. In simple terms, one can estimate the required sampling frequency based on the anticipated change in pollutant concentration or load, or turn the analysis around and estimate the change in pollutant concentration or load that is needed for detection with a monitoring design at a specified sampling frequency. The basic steps for conducting MDC analysis and consideration of matters such as the availability of representative data, the distribution of available data, independence of data values, the need for data transformation, and level of statistical significance are touched upon lightly here, but described and illustrated in detail in Spooner et al. (2011).

Sampling frequency determination is very closely related to MDC calculations. Sample size determination is usually performed by fixing a significance level, power of the test, the minimum change one wants to detect, the duration of monitoring, and the type of statistical test. MDC is calculated similarly, except that the sample size (i.e., number of samples), significance level, and power are fixed and the minimum detectable change is computed. In short, MDC is the amount of change you can detect given the sample variability.

3.4.2.2 Steps to Calculate the MDC

The calculation of MDC or the water quality concentration change required to detect significant trends requires several steps described by Spooner et al. (1987 and 1988) for a power of 50 percent. This general procedure varies slightly based upon:

- Whether the appropriate statistical model assumes a step or linear trend.
- Whether the data used are on the original scale (e.g., mg/L or kg) or log transformed.
- Incorporation of time series to adjust for autocorrelation.
- Addition of explanatory variables such as streamflow or season.
- Whether an alternative power is selected.

The following assumptions are made in the calculation of MDC.

- Historical sample measurements are representative of the temporal and spatial variation of the past and future conditions.
- Variability due to sampling, transport or laboratory error is negligible compared to variability over time.

3.4.2.2.1 Step 1. Define the Monitoring Goal and Choose the Appropriate Statistical Trend Test Approach.

One goal may be to detect a statistically significant linear trend in the annual mean (geometric mean is using log transformed data) pollutant concentrations that may be related to land treatment changes. A linear regression model using log-transformed data would be appropriate. An alternative goal to detect a statistically significant change in the post-BMP period as compared to a pre-BMP period would require a step change statistical test such as the *t*-test or ANCOVA.

3.4.2.2.2 Step 2. Exploratory Data Analyses.

The water quality data sets are examined to verify distributional assumptions required for parametric statistical procedures. Specific attention is given to the statistics on normality, skewness, and kurtosis. Preliminary data inspections are used to determine if the residuals follow a normal distribution with constant variance, both of which are required for the parametric analyses to be used. Both the original and logarithmic transformed values are tested. See section 7.10 for a list of available software packages. Options for exploratory data analysis (EDA) include Minitab Basic Statistics (Minitab 2016) and the SAS procedure PROC UNIVARIATE (SAS Institute 2012).

3.4.2.2.3 Step 3. Data Transformations.

Water quality data often follow log-normal distributions. In these cases, use the base 10 logarithmic transformation for the dependent variables (e.g., TP) to minimize the violation of the assumptions of normality and constant variance. Explanatory variables in statistical trend models do not have any distributional requirements because it is only the distribution of the residuals that is crucial. However, if they do exhibit log normal distribution, exploratory variables (e.g., upstream concentrations, flow) are also log-transformed which usually helps with the distribution of the residuals. When log transformation is required for the dependent variables, the log-transformed data are used in all MDC calculations leading to Step 7.

3.4.2.2.4 Step 4. Test for Autocorrelation.

Perform tests for autocorrelation on the water quality time series. An autoregressive, lag-1 (AR(1)) structure in biweekly or weekly samples is common. The tests usually assume samples are collected with equal time intervals. Methods to test for autocorrelation are described in detail in section 7.3.6.

3.4.2.2.5 Step 5. Calculate the Estimated Standard Error.

The variability observed in the historic or pre-BMP water quality monitoring data is used to calculate the MDC estimate. The estimated standard error is obtained from using the same statistical model selected in Step 1.

For a linear trend, use regression models with a linear trend, time series errors, and other optional explanatory variables to obtain an estimate of the standard deviation on the slope over time. If adjusting for autocorrelation, use a software procedure such as SAS's PROC AUTOREG to get the correct standard error on the slope. Alternatively, you can use the standard error adjustment for autocorrelated data given below in this step. For a step trend, use a *t*-test or ANCOVA with appropriate time series and explanatory variables to estimate the standard deviation of the difference between the mean values of the pre-BMP vs. post-BMP data ($s_{(\bar{X}_{pre}-\bar{X}_{post})}$). In practice, an estimate is obtained by using the following formula:

$$s_{(\bar{X}_{pre}-\bar{X}_{post})} = \sqrt{\frac{MSE}{n_{pre}} + \frac{MSE}{n_{post}}}$$

Where: $n_{pre} + n_{post}$ = the combined number of samples in the pre- and post-BMP periods

$s_{(\bar{x}_{pre}+\bar{x}_{post})}$= estimated standard error of the difference between the mean values in the pre- and the post- BMP periods.

MSE= s_p^2 = Estimate of the pooled Mean Square Error (MSE) or, equivalently, weighted average ("pooled") of the variances within each period. The MSE estimate is obtained from the output of a statistical analysis using a *t*-test or ANCOVA with appropriate time series and explanatory variables. If post-BMP data are not available, no autocorrelation is present, and no explanatory variables are appropriate (i.e., the simplest case), MSE or s_p^2 can be estimated by the variance (square of the standard deviation) of pre-BMP data.

The standard error on the trend estimate for simple trend models (e.g., step, linear, or ramp trends) with AR(1) error terms is **larger** than that (incorrectly) calculated by software procedures that do not include a correction for autocorrelation. The following adjustment can be applied to obtain the correct standard error for weekly or biweekly water quality data (Matalas, 1967; see Spooner et al. 2011 for additional details):

$$std.dev._{corrected} = std.dev._{uncorrected} \sqrt{\frac{1+\rho}{1-\rho}}$$

Where: $std.dev_{corrected}$ = true standard deviation of the trend (slope or difference between 2 means) estimate

$std.dev_{uncorrected}$ = incorrect standard deviation of the trend estimate calculated without regard to autocorrelation

ρ = autocorrelation coefficient for autoregressive lag 1, AR(1)

3.4.2.2.6 Step 6. Calculate the MDC.

For a power of 50 percent, the MDC is essentially one-half of the confidence interval for the slope of a linear regression trend or for the step trend difference between the mean values of the pre-and post-BMP periods. For a linear trend, the MDC is equal to one-half of the confidence interval on the slope obtained by multiplying the estimate standard deviation of the slope by the *t*-statistic, the total monitoring timeframe, and a correction factor for the additional planned monitoring years (see Spooner et al. 2011 for formulas). For a step trend, the MDC is one-half of the confidence interval to detect a change between the mean values in the pre- vs. post- BMP periods.

3.4.2.2.7 Step 7. Express MDC as a Percent Change.

If the data analyzed were not log-transformed, this is just the MDC divided by the average values in the pre-BMP period expressed as a percentage. If the data were log-transformed, a simple calculation can be performed to express the MDC as a percent decrease in the geometric mean concentration relative to the initial geometric mean concentration or load. The calculation is (see details and examples below):

$$\text{MDC\%} = \left(1 - 10^{-\text{MDC}'}\right) \times 100$$

where MDC′ is the MDC on the log scale and MDC% is a percentage.

3.4.2.3 Examples

The simplest example of an MDC calculation assumes a step trend, no autocorrelation, no covariates or explanatory variables, and Y values on the original scale (i.e., not transformed); see Spooner et al. (2011) for examples of linear trends with autocorrelation and covariates, as well as a paired watershed study or above/below-before/after studies. In this simple example, the planned comparison would be to detect a significant change in the average values between the pre- and post-BMP periods. The pre- and post-periods can have different sample sizes but should have the same sample frequency. Note: in this simplified example, the MDC would be equivalent to the Least Significant Difference (LSD) and would be calculated with a power of 50 percent as:

$$\text{MDC} = t_{(n_{pre}+n_{post}-2)} * s_{(X_{pre}+X_{post})}$$

Or, equivalently:

$$\text{MDC} = t_{(n_{pre}+n_{post}-2)} \sqrt{\frac{\text{MSE}}{n_{pre}} + \frac{\text{MSE}}{n_{post}}}$$

Where: $t_{(n_{pre}+n_{post}-2)}$ = one-sided[2] Student's *t*-value with (n_{pre} + n_{post} -2) degrees of freedom.

n_{pre} + n_{post} = the combined number of samples in the pre- and post-BMP periods

$s_{(\bar{x}_{pre}+\bar{x}_{post})}$= estimated standard error of the difference between the mean values in the pre- and the post- BMP periods.

MSE= s_p^2 = Estimate of the pooled Mean Square Error (MSE)

[2] The choice of one- or two-sided *t*-statistic is based upon the question being asked. Typically, the question is whether there has been a statistically significant decrease in pollutant loads or concentrations and a one-sided *t*-statistic would be appropriate. A two-sided *t*-statistic would be appropriate if the question being evaluated is whether a change in pollutant loads or concentrations has occurred. The value of the *t*-statistic for a two-sided test is larger, resulting in a larger MDC value.

Calculation Example #1 (post-BMP data not available): It is assumed that there will be two years of pre-BMP monitoring following by five years of post-BMP monitoring. For this example calculation, we assume bi-weekly sampling to avoid serious autocorrelation concerns and the need for adjustment. Example #2 illustrates an approach to address autocorrelation associated with weekly sampling.

n_{pre} = 26 samples/yr × 2 yr = 52 in the pre-BMP period
n_{post} = 26 samples/yr × 5 yr = 130 in the post-BMP period
Mean X = 36.9 mg/l, mean of the 52 samples in the pre-BMP period
s_p = 21.2 mg/L = standard deviation of the 52 pre-BMP samples
MSE= s_p^2 = 449.44
$t_{(n_{pre}+n_{post}-2)} = t_{180}$ = 1.6534 (one-sided)

The MDC would be:

$$\text{MDC} = t_{(n_{pre}+n_{post}-2)}\sqrt{\frac{\text{MSE}}{n_{pre}}+\frac{\text{MSE}}{n_{post}}}$$

$$\text{MDC} = 1.6534\sqrt{\frac{449}{52}+\frac{449}{130}}$$

$$\text{MDC} = 5.7 \text{ mg/l}$$

Percent change required = 100 × (5.7/36.9) = 15%

So, in this example, sampling bi-weekly before (2 years) and after (5 years) BMP implementation would require a 15 percent change in concentration to be detectable at the 95 percent confidence level and a 50 percent power. If a smaller change was anticipated, then sampling frequency (or duration) would need to be increased to adjust for autocorrelation. If a decrease of more than 15 percent was expected, then sampling frequency could be decreased and the MDC recalculated to determine if the reduced sampling frequency would be adequate to meet project goals. Because MDC analysis is used to "estimate" detectable change it is recommended that estimated sampling frequency needs are assumed to be higher than calculated to reduce the risk of failure.

Calculation Example #2 (post-BMP data not available, similar data distribution as in example #1): It is assumed that there will be two years of pre-BMP monitoring following by five years of post-BMP monitoring. For this example

Autocorrelation

Essentially means that subsequent samples are influenced by previous samples. These subsequent samples contain less new information than would otherwise be obtained from a completely independent additional sample (i.e., there is information overlap). The result is that autocorrelation reduces the *effective* sample size compared to the situation with no autocorrelation.

Rho**(ρ)** is the **coefficient of autocorrelation**, and basically describes the relationship between the current and its past values.

- Rho increases as the strength of the relationship between current and past samples increases.
- Larger rho means that each collected sample has less new information (i.e., effective sample size is reduced).
- So, the relative improvement in estimates of a mean or a minimum detectable change decreases as sample size increases.
- Rho is used to adjust the standard deviation for inclusion in the step-change MDC calculations demonstrated in this section.

calculation, we assume weekly sampling and address autocorrelation by assuming an autocorrelation coefficient of ρ=0.3 (common for NPS projects with weekly sampling). A corrected standard deviation is calculated as (see Spooner et al. 2011 for additional details):

$$pooled\ std.dev._{corrected} = 21.2 \times \sqrt{\frac{1+\rho}{1-\rho}} = 21.2\ x \sqrt{\frac{1+0.3}{1-0.3}} = 28.9$$

Therefore:

n_{pre} = 52 samples/yr × 2 yr = 104 in the pre-BMP period

n_{post} = 52 samples/yr × 5 yr = 260 in the post-BMP period

Mean X = 36.9 mg/l, mean of the 52 samples in the pre-BMP period

s_p = 28.9 mg/L = corrected standard deviation of the 52 pre-BMP samples

MSE= s_p^2 = 834.67

$t_{(n_{pre}+n_{post}-2)}$ = t_{362} = 1.6491 (one-sided)

The MDC would be:

$$MDC = t_{(n_{pre}+n_{post}-2)} \sqrt{\frac{MSE}{n_{pre}} + \frac{MSE}{n_{post}}}$$

$$MDC = 1.6491 \sqrt{\frac{835}{104} + \frac{835}{260}}$$

$$MDC = 5.5\ mg/l$$

Percent change required = 100 × (5.5/36.9) = 15%

So, in this example, sampling weekly before (2 years) and after (5 years) BMP implementation would also require at least a 15 percent change in concentration to be detectable at the 95 percent confidence level and a 50 percent power. In essence, autocorrelation results in diminishing returns for higher sample frequencies. However, it should be noted that even biweekly sample frequency such as used example #1 also have autocorrelation, just a lesser amount (e.g., ρ=0.1) which would have resulted in a MDC estimate for biweekly sampling in example #1 of 17.2 percent.

3.4.2.4 Factors Affecting the Magnitude of the MDC

Up to this point the discussion of MDC analysis has been based on simplifying assumptions. The reality, however, is that the true MDC value for a specific significance level varies as a function of pollutant variability, sampling frequency, length of monitoring time, other factors (e.g., potential explanatory variables such as season, meteorological, and hydrologic variables), the magnitude and structure of the autocorrelation (see Calculation Example #2 above), and the statistical techniques used to analyze the data. Variations in water quality measurements are due to several factors including:

- A change in land treatment or land use resulting in decreased (hopefully) concentrations and/or loadings to receiving waters (determining the amount of water quality change is usually a key objective of a watershed project).
- Sampling and analytical error.
- Monitoring design (e.g., sampling frequency, sampling location, variables measured).
- Changes in meteorological and hydrologic conditions.
- Seasonality.
- Changes in input to and exports from the system. For example, changes in upstream concentrations can affect the downstream water quality.

The bottom line is that the magnitude of MDC is often larger than expected but can be reduced by:

- Accounting for changes in discharge, precipitation, ground water table depth or other applicable hydrologic/meteorological explanatory variable(s).
- Accounting for changes in incoming pollutant concentrations upstream of the BMP implementation subwatershed (i.e., upstream concentrations).
- Increasing the length of the monitoring period.
- Increasing the sample frequency.
- Applying the statistical trend technique that best matches the implementation of BMPs and other land use changes.

Figure 3-21 through Figure 3-24 illustrate how MDC varies with sampling frequency/duration, confidence level (expressed as percent), coefficient of variation (CV), and autocorrelation coefficient (ρ), respectively using a 50 percent power. These examples all assume a step trend and no covariates or explanatory variables and use the basic equation found in section 3.4.2.3. The CV is used in lieu of standard deviation because it has broader applicability (CV=std.dev./mean). Figure 3-22 to Figure 3-24 assume a seven-year monitoring program (two pre-BMP and five post-BMP) with the same sampling frequency each year. Data are assumed to follow a normal distribution and pre- and post-BMP CVs are assumed to be the same. In Figure 3-21 through Figure 3-23, the values of ρ were assumed to be 0.1 and 0.3 for sampling frequencies of 26 and 52 times per year, respectively. No autocorrelation was assumed for less frequent sampling.

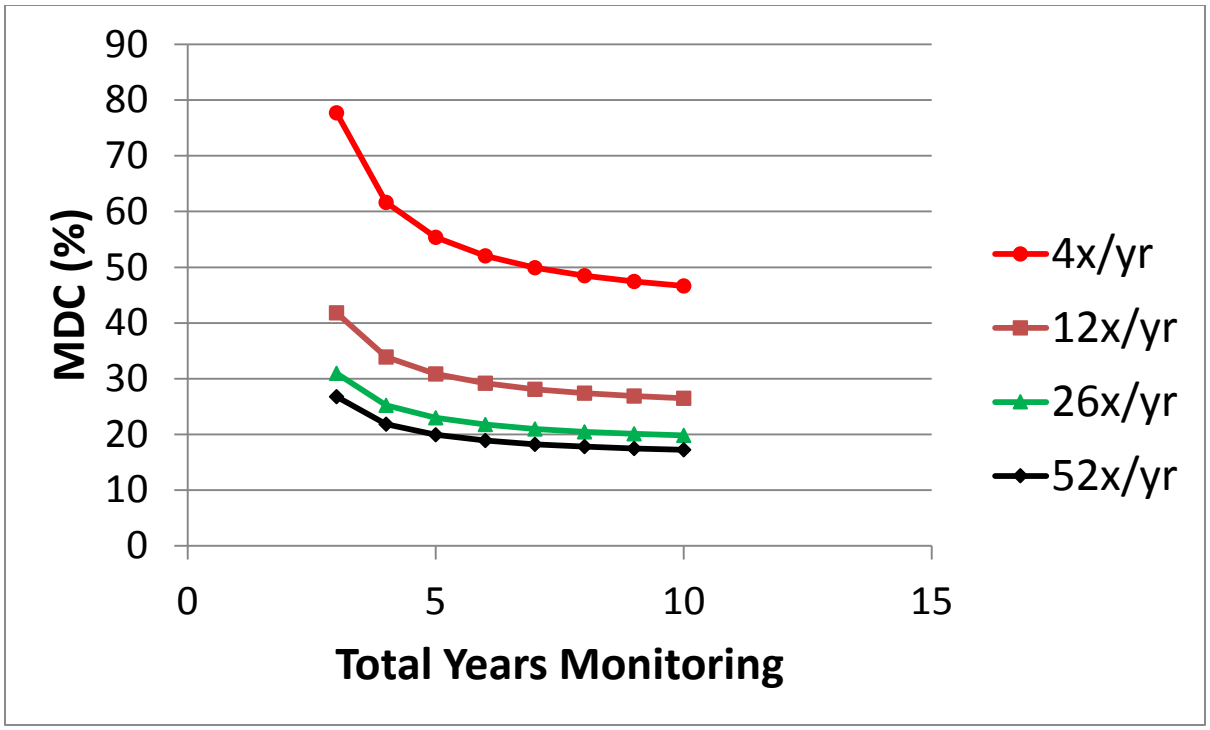


Figure 3-21. MDC versus frequency and years of monitoring. Assumes ρ=0.1 for 26x/yr and 0.3 for 52x/yr, CV=0.7, and 95% confidence level.

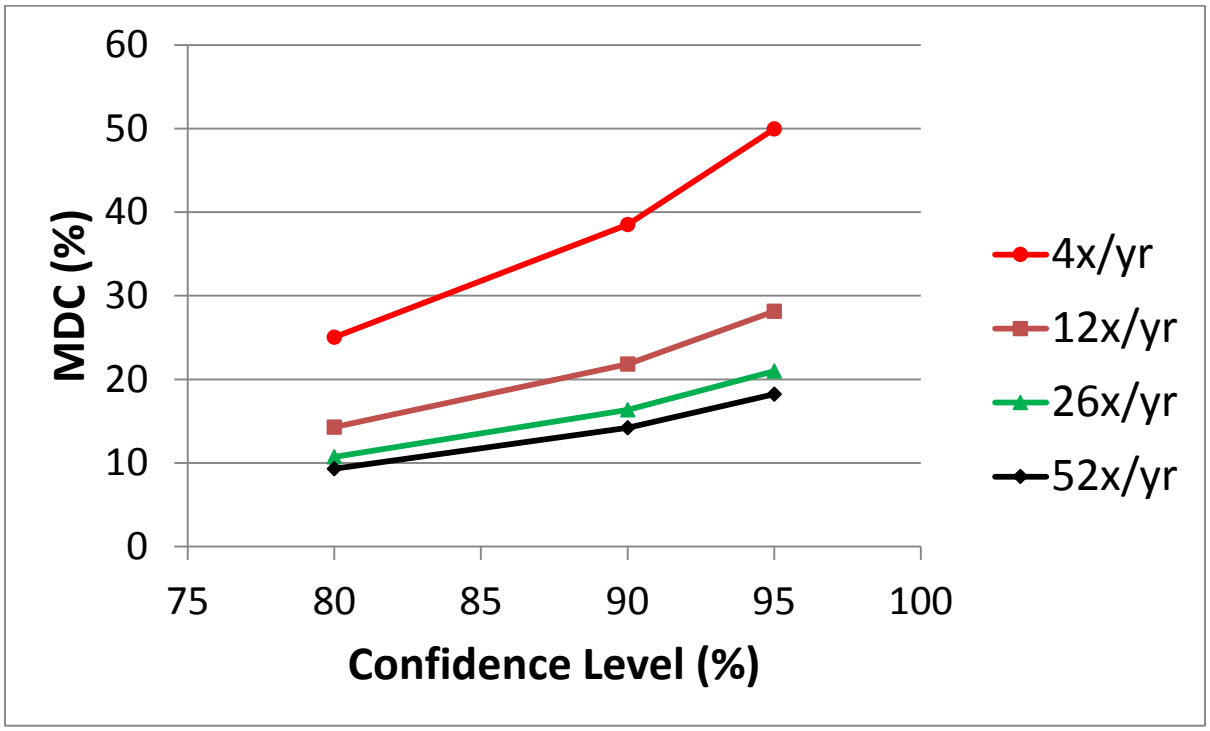


Figure 3-22. MDC versus confidence level. Assumes ρ=0.1 for 26x/yr and 0.3 for 52x/yr, 7 years of monitoring, and CV=0.7.

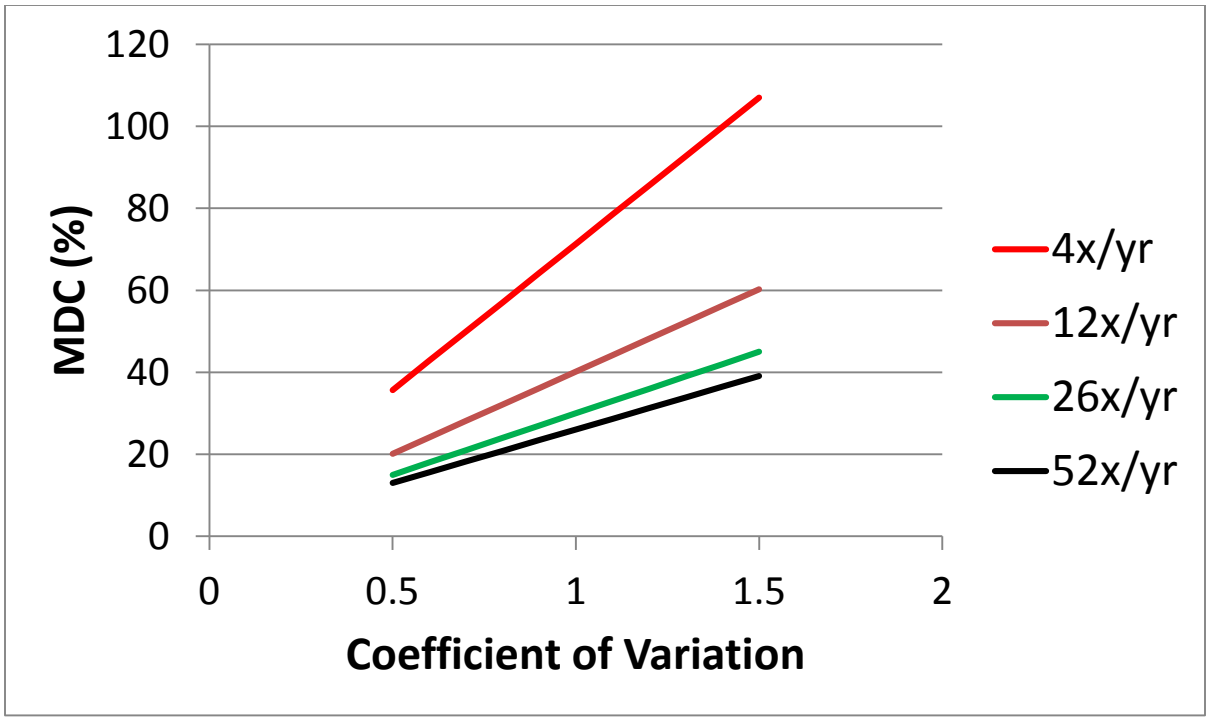


Figure 3-23. MDC versus coefficient of variation. CV calculated using unadjusted std. dev. Assumes ρ=0.1 for 26x/yr and 0.3 for 52x/yr, 7 years of monitoring, and 95% confidence level.

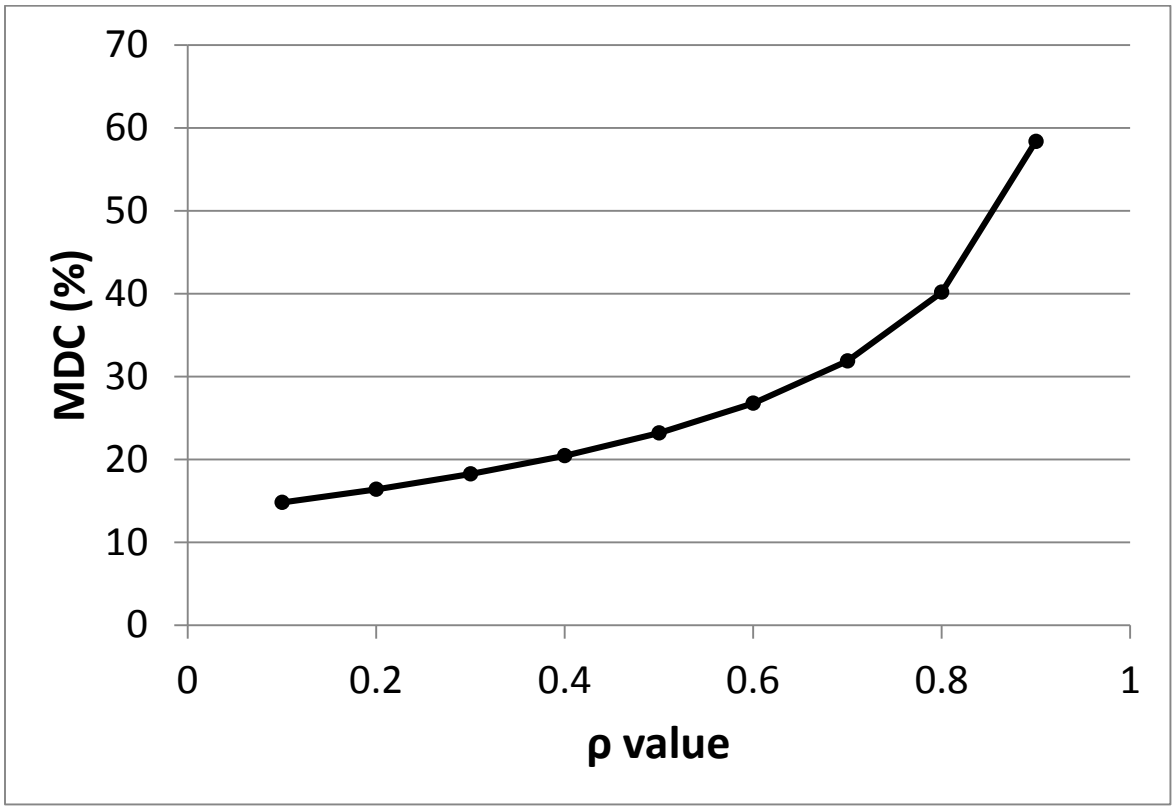


Figure 3-24. MDC versus coefficient of autocorrelation (ρ). Assumes 7 years of monitoring, 52x/yr, CV=0.7, and 95% confidence level. MDC = 13% if no autocorrelation is assumed.

Figure 3-21 shows that the change in MDC is less pronounced with increasing duration for designs with more frequent sampling. For example, MDC drops from 78 percent to about 62 percent when monitoring is extended from three to four years with quarterly sampling; the corresponding change for weekly sampling is only 5 percent. In addition, the change in MDC is minor after seven years for monthly or more frequent sampling. Figure 3-22 illustrates the benefits of considering the statistical confidence needed in changes that might be documented. Sampling 26 times/year over seven years, the MDC drops from 21 to 11 percent when the confidence level is changed from 95 to 80 percent, respectively. In some cases management decisions can be based on less than 95 percent confidence. These changes are more pronounced at lower sampling frequencies. Figure 3-23 illustrates the importance of having a good estimate of variance when calculating MDC. An assumption that the CV=0.5 when it is actually 1.5 could result in a monitoring plan designed to detect an MDC of 15 percent at 26 samples per year when the actual MDC is 45 percent. Finally, Figure 3-24 illustrates the impact of autocorrelation on MDC estimates. In this example (52 samples/year for seven years), the MDC increases with increasing autocorrelation, with an MDC of 20 percent at ρ=0.4 and an MDC of 32 percent at ρ=0.7. Testing for autocorrelation is an important element of using existing data to aid in monitoring plan development, particularly when anticipated sampling frequencies exceed about 25 or more per year.

The reader is referred to Spooner et al. (2011) for additional details on estimation of MDC.

3.4.3 Sampling Duration

How long should a monitoring program be conducted? The answer is essentially: as long as needed to achieve the objectives or document a change. Following are basic guidelines for ensuring that a planned monitoring program has a reasonable chance of success.

- **Capture at least one full cycle of natural or cultural variability.** Especially for NPS situations, monitoring should be conducted long enough to capture the full range of expected variability from weather, seasons and cultural factors such as cropping patterns or construction management. Similarly, if the first year of monitoring is done in a notable drought period, it would be wise to extend monitoring to capture a more representative set of weather conditions.
- **Use statistical tests to evaluate the adequacy of a monitoring period.** Data from some monitoring designs can be tested statistically to determine if an adequate database exists. For example, data from a paired-watershed design (see section 2.4.2.8) can be tested to determine if acceptable calibration has been achieved and if treatment can begin (USEPA 1993b). Pre-treatment data from a before/after design can be evaluated for MDC to help determine if it is likely that enough data exist to document an expected change.
- **Consider lag time.** Lag time between land treatment and water quality response is a common phenomenon (see section 6.2). Knowledge of key lag time factors can help determine the required duration of a monitoring program. For example, if groundwater travel time from an agricultural field through a riparian forest buffer to a stream is known to be five to 10 years, it is reasonable to expect to continue monitoring at least that long. Similarly, a lake with a flushing rate of 1.5 years may respond much more quickly to changes in pollutant inputs and a shorter monitoring program could suffice.

3.5 Monitoring Station Construction and Operation

This section discusses the design and operation of physical facilities involved in fixed monitoring stations. The type of station required depends on both project objectives and the nature of the resource

being monitored. Not all monitoring designs require fixed station facilities, e.g., synoptic/grab sampling, lake monitoring, biological monitoring. When physical facilities are required, several important principles apply, regardless of station type.

- **Select monitoring sites according to specific criteria based on program objectives and needs** (see section 3.3).
- **Design the station to collect representative samples from the target population under foreseeable circumstances.** Make certain that measurements and samples are taken from areas that represent the resource or problem of interest, e.g., from the main flow of a stream, not an eddy; from a well-mixed area below a discharge; from the geologic formation transmitting subsurface flow. In situations where vertical or horizontal variability exists, depth-integrated samples or several discrete samples may be required. Physical facilities should allow access and sample collection during anticipated high flows, harsh climates, or inclement weather.
- **Strive for simplicity.** While sophisticated technology offers many capabilities and advantages, power failures and unexpected errors may occur and cause problems in complex designs. When possible, the simple alternative may well be the best choice. A passive crest gage may provide necessary information on peak stream stage more reliably than an electronic sensor. In addition, monitoring systems with data loggers and real-time internet uplinks may function well most of the time, but there is often no substitute for a regular visit by a field technician to maintain equipment and to record key data and observations.
- **Include redundancy.** When possible, provide a backup means of collecting essential samples or data. This may mean including a passive sampling device like a US U-59 single stage sampler (Wilde et al. 2014) as a backup to an autosampler. A flow totalizer on a flow meter provides data on total event discharge in case a data logger fails or a file is corrupted and the continuous stage and flow data are lost.
- **Provide security.** Monitoring instruments and equipment need to be protected both from the elements and from potential vandalism. Field technicians need safe access and protection from inclement weather and other hazards. The integrity of samples and accumulated data should be protected so that adequate chain of custody is maintained.

The following sections discuss important aspects of monitoring station design for several common applications including streams and rivers, edge of field, and individual structures or BMPs. The following are examples of comprehensive references that provide additional detail on these and other matters of monitoring station design.

- USDA *Field Manual for Research in Agricultural Hydrology* (Brakensiek et al. 1979)
- USDA-NRCS *National Handbook of Water Quality Monitoring* (USDA-NRCS 2003)
- USGS *National Field Manual for the Collection of Water Quality Data* (USGS variously dated)

When selecting specific instrumentation and equipment for monitoring stations, review manufacturer information for features and specifications to be sure that equipment can do the jobs required.

3.5.1 Grab Sampling

Even though monitoring programs based exclusively on grab sampling may not require “stations” with physical facilities, grab sampling stations must be located and identified so that samples can be repeatedly collected from the same location. Such locations may be fairly obvious such as road crossings on streams

or pipes delivering flow to or from a stormwater treatment system. These sampling locations can simply be recorded on a map or in a standard operating procedure. In lakes, however, repeated navigation to a specific location will likely require use of a global positioning system (GPS) device. Determination of sampling depth will also be required at some lake stations, using a weighted line or an electronic depth sounder. There are numerous devices available to collect grab samples. The choice will depend on water resource characteristics, the type of sample desired (e.g., surface vs. depth-integrated), and on the variable(s) to be monitored (see section 3.6.2.1).

3.5.2 Perennial Streams and Rivers

Long-term stations to continuously record streamflow and collect periodic water samples require structures and facilities to house monitoring equipment. Specific considerations for flow measurement have been discussed previously (see sections 3.1.3.1 and 3.3.2.2). Stations for continuous flow measurement require a staff gage and a means of continuously recording stage, e.g., using a stilling well with a float or bubbler or directly in the channel using a bubbler, pressure transducer, or ultrasonic device. The traditional float gage in a stilling well is highly reliable and is protected from turbulence, ice and debris in the stream channel. Advantages of bubblers, transducers or ultrasonic devices are they can be placed directly in the stream channel, data can be logged electronically, and flow data can be linked to an autosampler. A diagram of a stream station with an in-stream pressure transducer and staff gages is shown in Figure 3-25 (Freeman et al. 2004).

Water samples at continuous monitoring stations are typically collected by autosamplers. Autosamplers commonly pump samples from the stream through plastic tubing and collect the water in one or more bottles. Modern autosamplers are sophisticated instruments that can collect timed samples of specific volume based on their own internal programs or collect storm-event or flow-proportional samples when linked to a flow recorder or other triggering device (see section 3.6.2.4). One common issue associated with pumping autosamplers is the nature and placement of the intake. Sampler intake is usually fixed at some point in the stream and may not collect a sample representative of vertical or horizontal variability. Some depth-integrated intake devices have been proposed and tested with success (Eads and Thomas 1983), but some of these devices can require frequent maintenance and can be impractical in northern climates where ice is a problem. Selbig and Bannerman (2011), however, demonstrated the idea of vertical stratification of solids in storm sewer runoff using a fully-automated, depth-integrated sample arm (DISA) for collecting integrated samples within pipes (Figure 3-26). Subsequent laboratory testing showed that the DISA was better able to characterize suspended-sediment concentration and particle size distribution compared to fixed-point methods (Selbig et al. 2012).

Some variables (like temperature, turbidity, specific conductance, and dissolved oxygen) can be monitored *in situ* without collecting actual water samples using sensors deployed directly in the stream. Installation and operation of such sensors for continuous monitoring requires consideration of site-specific characteristics related to exposure of the sensors to the water, mounting platforms, protection from fouling and impact from debris, calibration, and maintenance. Consult manufacturer recommendations and additional resources for specific guidance on sensors (e.g., Miles 2009, USEPA 2005b).

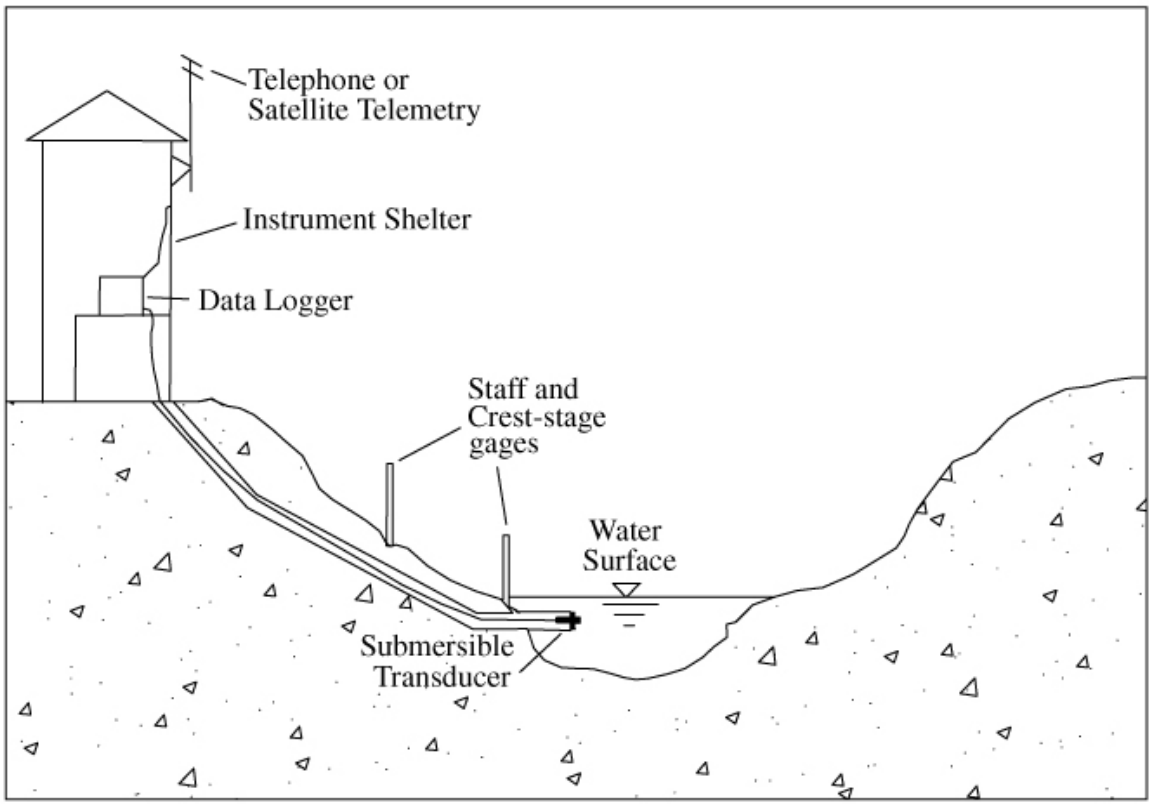


Figure 3-25. Monitoring station with submersible transducer in stream (Freeman et al. 2004)

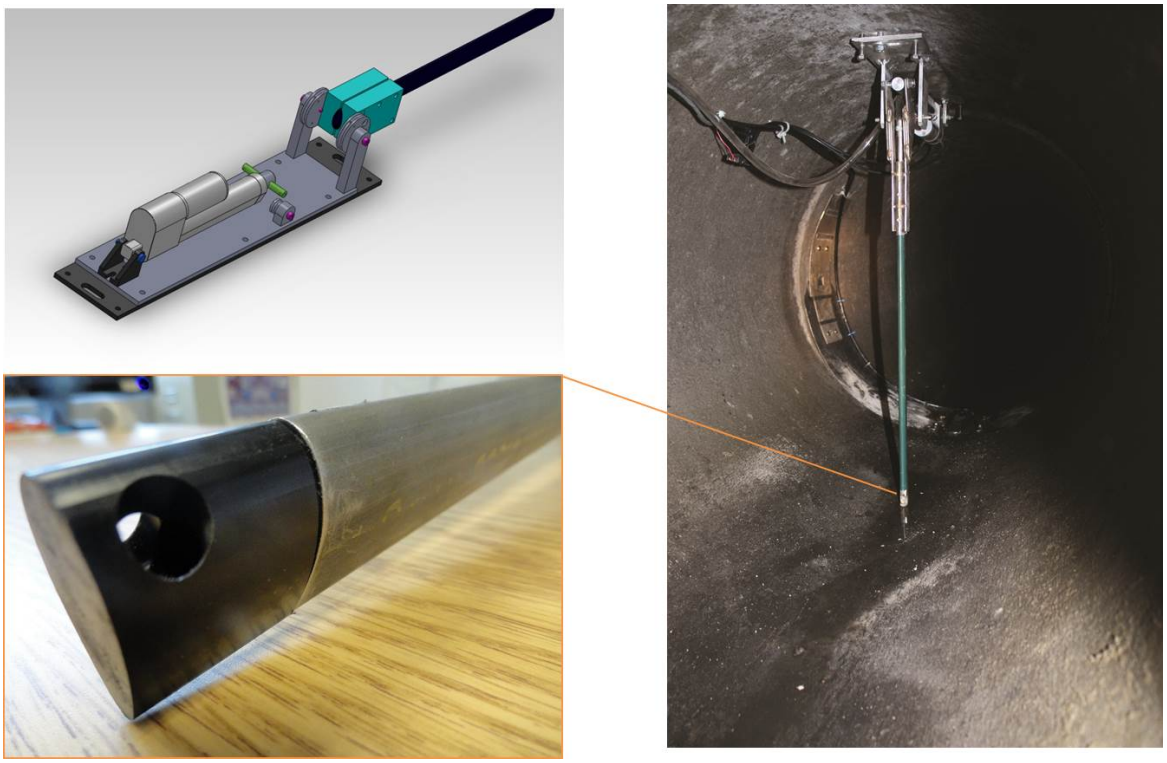


Figure 3-26. Drawing and field installation of depth-integrated sample arm for automatic samplers (photo by R.T. Bannerman, Wisconsin DNR)

The major advantage of autosamplers and recording sensors is that they can operate unattended for extended periods. Autosamplers, for example, can remain dormant for weeks and triggered by precipitation or rising flow independent of personnel action. This is particularly important when monitoring transient storm events is an objective. However, such equipment is expensive and requires regular maintenance and calibration.

Modern monitoring instruments can be linked together with a data logger (either a separate unit or part of either the flow meter or autosampler) for sampling control and data storage. Where resources are available, stations can be equipped to communicate through cell phone systems or Internet in real time. In such cases data can be downloaded and commands for sampling or recording data can be sent remotely. If this kind of system is used, issues of communication linkage such as line-of-sight for radio or connectivity for cell phones must be considered during station design.

Unless a completely passive, mechanical system is devised, most water quality monitoring stations will require electrical power. Power can be provided with deep-cycle automotive or marine batteries, but servicing and recharging batteries may be problematic and battery power may be inadequate for running refrigeration or heating. For long-term application, it is desirable to obtain AC power from either the electrical grid or a properly designed solar charging system. It should be cautioned, however, that electronic monitoring instruments are often vulnerable to voltage spikes that may occur, especially in rural areas, and computer-type power surge protectors should be used to prevent instrument damage.

Finally, it should be noted that stream monitoring stations face a number of challenges in northern climates. Ice in the stream channel can disrupt a stage-discharge rating (see section 3.1.3.1) and disable or destroy sampling lines or instruments located in the stream. Winter weather may require robust shelter and prolonged low temperatures may require heat from heating tape or propane heaters to prevent samples and equipment from freezing. Conversely, stations in hot climates may require special cooling and/or ventilation for proper operation. Such requirements must be considered in designing monitoring stations.

3.5.3 Edge of Field

"Edge of field" generally describes a situation where flow is intermittent and may or may not move through defined channels. For the purposes of this manual, this includes monitoring in waterways or points of concentrated flow at the edges of agricultural fields or in intermittent streams in any location associated with field drainage. Edge of field monitoring stations share many common requirements with stations on perennial streams, i.e., the need to measure flow (when it occurs), the need to collect representative water samples and other data, the need for power, and challenges of extreme weather. Edge-of-field stations face several additional challenges including:

- **Lack of a defined drainage channel,** requiring measures such as wingwalls or berms to direct flow into and/or out of the station.
- **Intermittent flow,** requiring that monitoring equipment be prepared for activation (e.g., by precipitation or flow) at any time.
- **Unpredictable timing and magnitude of flow,** requiring wide tolerances in flow and sampling capacity.
- **Remote location,** usually lacking easy access and power from the grid.

Stuntebeck et al. (2008) provides a comprehensive discussion of how these challenges were met in edge-of-field monitoring stations at the Discovery and Pioneer Farms in Wisconsin. Typical edge-of-field stations included these elements:

- **Enclosures** consisting of custom-made, aluminum, clam-style structure to house equipment designed to measure stage, collect water samples, and provide two-way telecommunication.
- **Stage and discharge equipment** including
  - Wingwalls and berms to collect overland flow.
  - A flume for discharge measurement.
  - A discharge outlet to prevent erosion and ensure proper flume operation.
  - A bubbler gage, pressure transducer, or acoustic sensor for water level recording.
  - A crest gage as a backup and calibration check for recorded stage data.
- **Sampling equipment** including an autosampler and sample intake line protected from freezing by using a down-gradient slope, heat tape, and foam insulation.
- **Data logging and control instruments.**
- **Communications** including radio modem and datalogging communications software.
- **Power** including solar-charged DC batteries for electronics operation and an AC generator for heating and sample refrigeration.
- **Digital time-lapse camera** to periodically record field conditions.

Finally, it should be noted that edge-of-field stations typically require more maintenance than continuous stream stations. Edge-of-field stations may have to remain dormant but ready for activation over extended periods between events, and regular maintenance visits are required even when inactive. This is particularly true in northern climates where removal of ice and snow in preparation for monitoring critical winter thaw or spring runoff events is especially labor-intensive.

Figure 3-27 shows examples of edge of field monitoring stations.

3.5.4 Structures/BMPs

Monitoring stations for specific BMPs or stormwater treatment structures are similar in many respects to edge-of-field stations, but require some additional considerations because of site characteristics and constraints.

Many individual BMP monitoring efforts have similar requirements for flow measurement, water sampling, data logging, communications, and security as other station types, but are often constrained by physical characteristics. Monitoring inflow and outflow from a constructed wetland is generally comparable to monitoring flow in an intermittent stream. Runoff from a parking lot entering an infiltration BMP, however, may be very difficult to quantify and sample, and outflow from the BMP may be carried in an underground pipe. Some specialized equipment for such monitoring has been developed, including passive runoff samplers (Figure 3-28) and flume inserts for pipes with integrated stage sensors (Figure 3-29). In a review of passive samplers for urban catchment studies, Brodie and Porter (2004) classified them based on the main hydraulic principle applied in their design: gravity flow, siphon flow, rotational flow, flow splitting, and direct sieving. In two Wisconsin studies, Parker and Busch (2013)

demonstrated the capabilities and limitations of a crown divisor sampler in the laboratory and at the edge of a small field, while Graczyk et al (2000) compared siphon samplers to automatic samplers in a stream setting. In urban settings, much of the monitoring equipment may need to fit into a catch basin or storm sewer access point. Station enclosures and security in urban areas may present additional challenges.

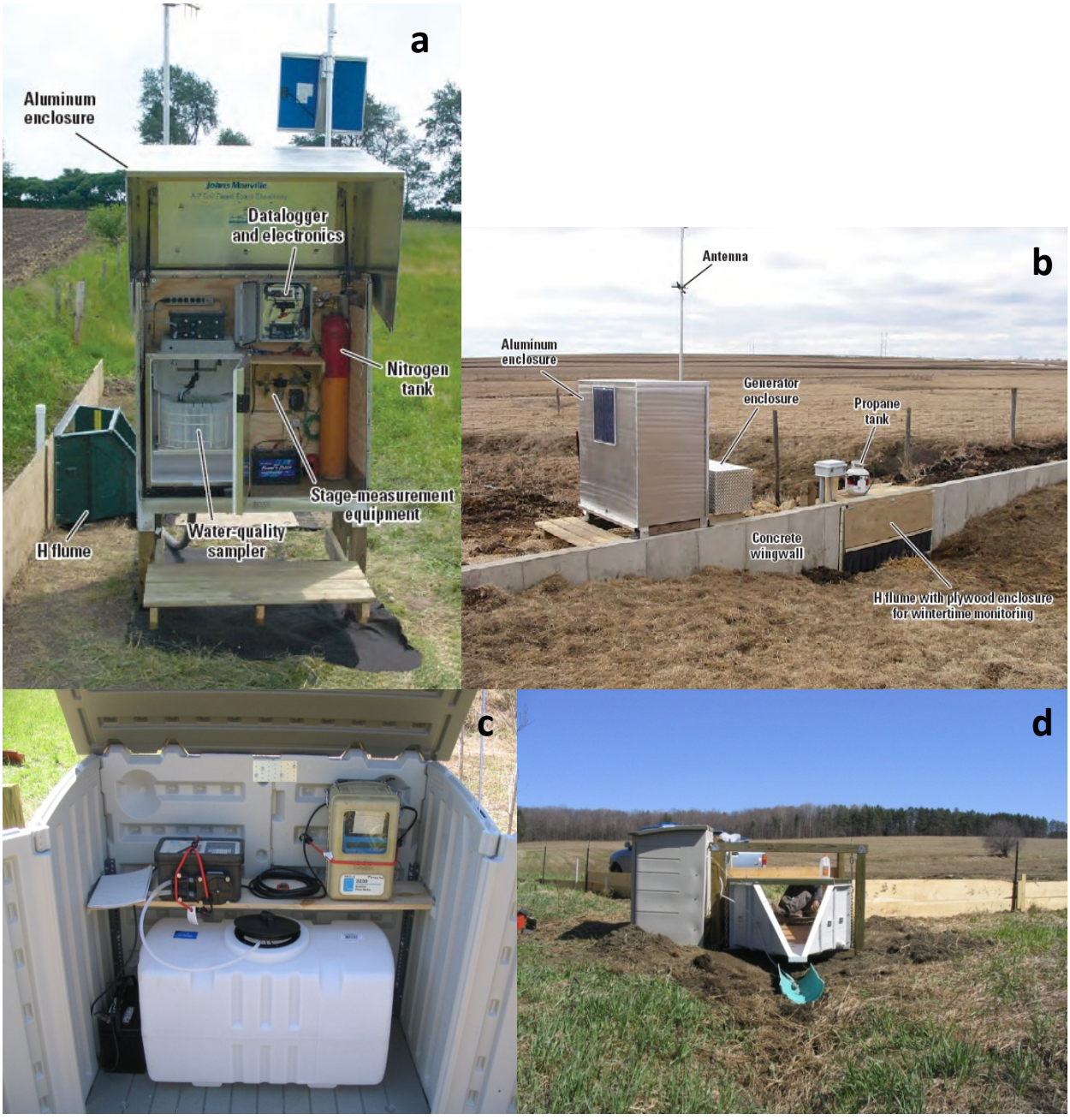
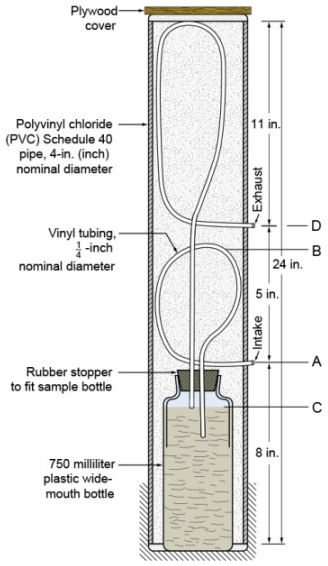
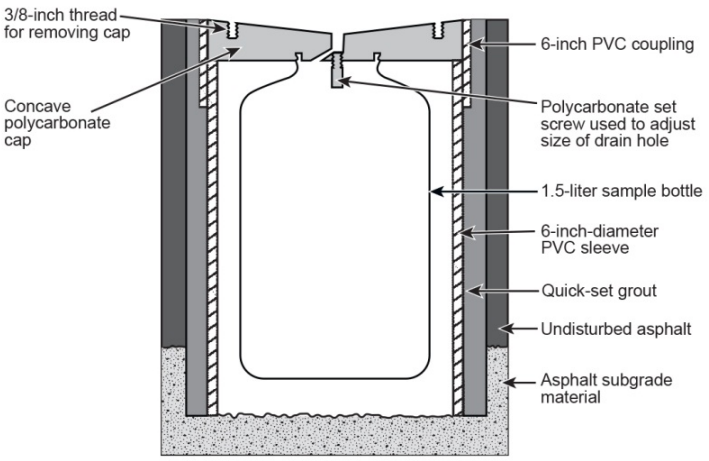


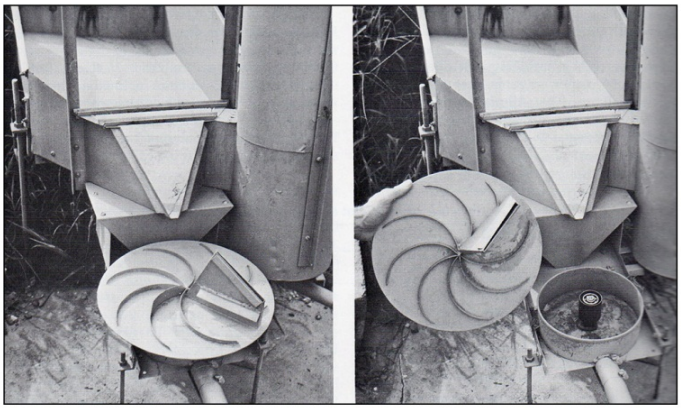
Figure 3-27. Edge-of-field monitoring stations. *a, b,* Wisconsin Discovery and Pioneer Farms (Stuntebeck et al. 2008); *c, d*, Vermont (Meals et al. 2011a).



A. Siphon Sampler



B. Street Runoff Sampler



C. Coshocton Wheel

D. Multi-slot Sampler

Figure 3-28. Examples of passive runoff samplers that can be used for edge-of-field or BMP studies (A-Graczyk et al. 2000, B-Waschbusch et al. 1999, C-Brakensiek et al. 1979, and D-Parker and Busch 2013; photo D by P. Parker, University of Wisconsin-Platteville)

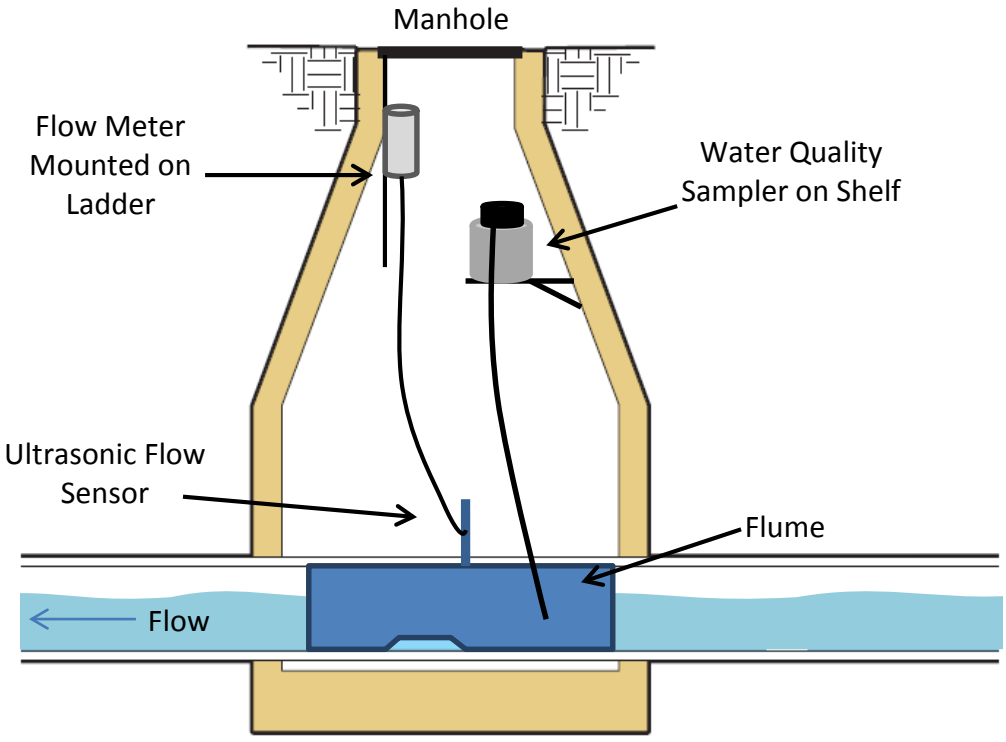
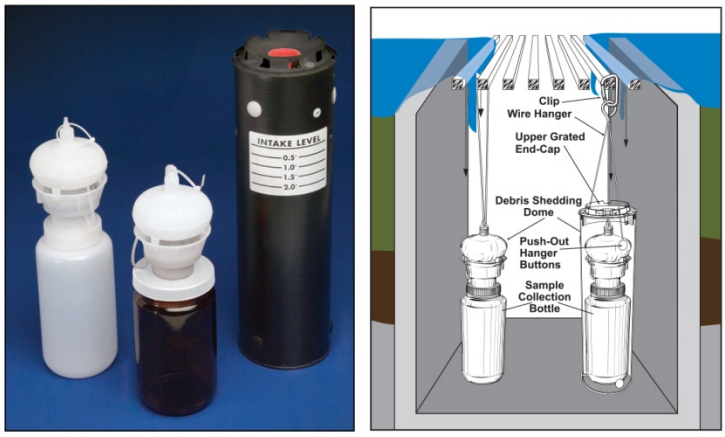


Figure 3-29. Flow measurement and water quality sampling in stormwater pipes

In urban runoff monitoring, the first flush phenomenon (the initial surface runoff from a rainstorm carrying high levels of pollutants that accumulated on impervious surfaces during dry weather) requires special consideration because pollutant loads during the first part of an event may be much larger than those in the later flows. Several approaches have evolved to monitor this phenomenon. Low-cost passive first-flush samplers are available that capture early surface runoff, then close when filled (Figure 3-30). Waschbusch et al. (1999) used a range of passive samplers to monitor street runoff, driveway runoff, lawn runoff (Figure 3-31), roof runoff, and parking lot runoff. Some modern autosamplers offer special settings for activation of intensive sampling programs at certain flow levels, then scale back sampling frequency later in the event (Figure 3-32).



A. Nalgene® first-flush sampler. Installed below grate (at right).

B. Edge-of-road sampler.

C. GKY first-flush sampler.

Figure 3-30. Examples of first-flush runoff samplers (A-Nalgene 2007, B-Barrett 2005, C-GKY 2014)

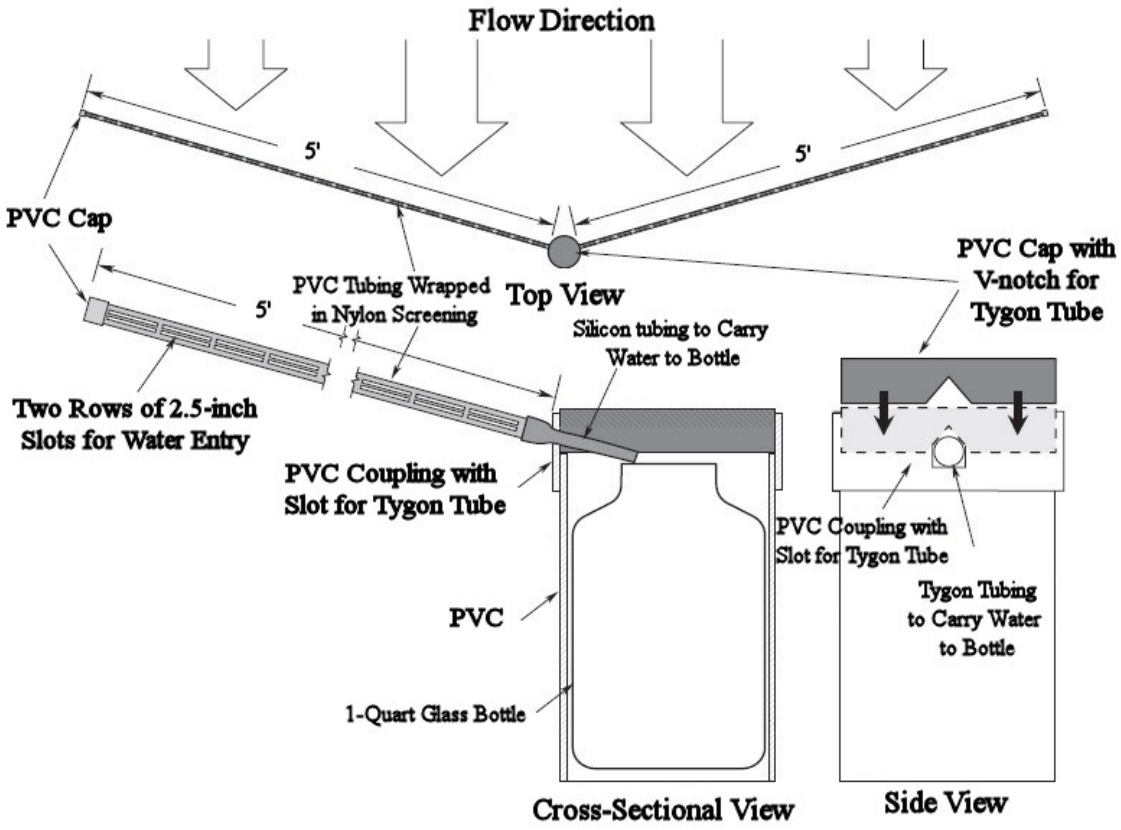


Figure 3-31. Passive sampling setup for lawn runoff (after Waschbusch et al. 1999)

Sigma 900 MAX Portable Standard Sampler

Isco 6712 Portable Sampler

Figure 3-32. Examples of automatic samplers with capabilities for variable sampling frequencies (Hach® 2013a, Teledyne Isco 2013a)

Many BMP monitoring efforts follow an input/output design, where water quality (i.e., concentration or load) is measured entering and again leaving the structure to assess pollutant reduction performance. Such cases not only require two monitoring stations but also require that the stations be coordinated so that water actually treated by the BMP is sampled properly. If sampling is conducted simultaneously at the entrance and exit of a stormwater BMP, for example, the outflow sample may represent "old" water pushed out of the BMP by "new" inflow, rather than new inflow after treatment by the BMP. Similarly, water quality measured simultaneously upstream and downstream of a feedlot may not reflect the influence of the feedlot, at least early in a storm event. Time of travel or residence time in the BMP must be considered in setting up monitoring stations. This can be accomplished by linking the above and below stations to better coordinate downstream with upstream sampling. Stuntebeck (1995), for example, modified the basic above/below design in a Wisconsin barnyard runoff study by setting the samplers to be activated by precipitation and programming them to collect time-integrated samples for an initial period. This modification allowed for sampling of barnyard runoff in the receiving stream before streamwater level increases could be sensed, thereby effectively isolating the barnyard runoff from nonpoint-pollution sources upstream. Secondly, this approach allowed sampling during small storms in which local inputs from the barnyard were apparent, but little storm runoff from the upstream areas of the watershed were observed. A second modification took advantage of the close proximity of the two stations to create a direct electronic connection between the stations for collection of concurrent samples.

3.5.5 Meteorology

Meteorological data, particularly precipitation data, are nearly always relevant to NPS monitoring projects (see section 3.1.5). The nature and extent of meteorological monitoring will vary according to monitoring objectives. Precipitation data are useful in driving event sampling and for documenting rainfall conditions relative to long-term averages. Particular monitoring objectives may require monitoring of other meteorological variables. A study of indicator bacteria runoff from agricultural fields, for example, may call for monitoring of weather conditions that influence bacteria survival in the field, such as air temperature, soil temperature, solar radiation, relative humidity, and wind velocity.

Guidance for meteorological monitoring is given in the USDA Agricultural Handbook 224 (Brakensiek et al. 1979) and in the National Weather Service Observing Handbook No. 2 (NWS 1989). Probably the most important criterion for precipitation measurement is location. For BMP or field monitoring efforts, a single meteorological station may be sufficient. For larger watershed monitoring, multiple stations are usually necessary to account for variation of weather with elevation, and other geographic factors. Multiple precipitation stations are especially important in monitoring efforts designed to provide data for model application. Successful application of watershed models such as SWAT is highly dependent on accurate precipitation data (Gassman et al. 2007). Precipitation monitoring stations must be located so that there are no obstructions within 45º of the lip of the gage (USDA-NRCS 2003). A more restrictive general rule, illustrated in Figure 3-33, indicates that an obstruction should not be closer to the gage than two to four times the obstruction's height above the gage (Brakensiek et al. 1979).

A variety of instrumentation is available for meteorological monitoring, including many electronic instruments that record directly into dataloggers. Tipping bucket rain gages measure both total accumulated rainfall and rainfall rate and can be connected to other monitoring instruments to log data and/or trigger sample collection. For winter operation, tipping bucket gages must be heated electrically. A weighing bucket precipitation gage can measure both rain and snow if it is charged with anti-freeze in the winter. It is generally a good idea to provide a manual (non-recording) rain gage on the station site as a backup and calibration check for the recording instrument.

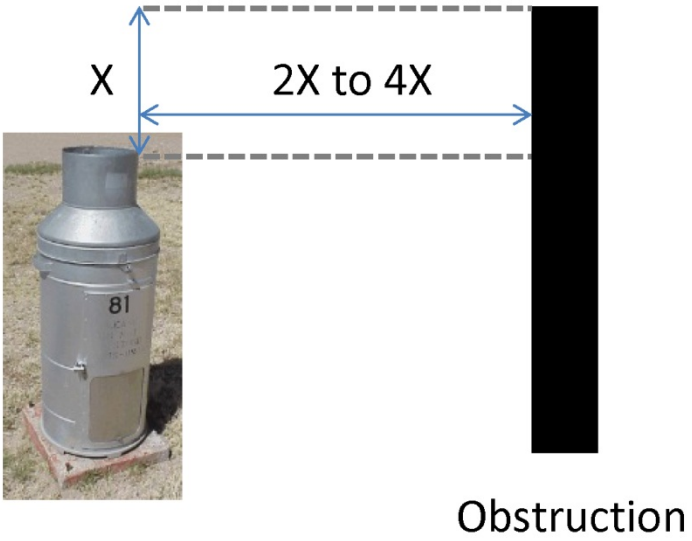


Figure 3-33. Precipitation gage placement relative to obstructions

An example of a meteorological station measuring precipitation, air temperature, solar radiation, relative humidity, and wind velocity is shown in Figure 3-34.

Figure 3-34. Photograph of a meteorological monitoring station (Meals et al. 2011a)

3.6 Sample Collection and Analysis Methods

Collection and analysis of samples, and obtaining measurements and other data from monitoring stations is an exacting task that requires training, appropriate equipment, careful adherence to standard procedures, and detailed record-keeping. This guidance discusses basic principles and important rules of thumb. Other sources such as the USGS National Field Manual for the Collection of Water Quality Data provide specific information and procedures.

3.6.1 General Considerations

This section presents some general aspects of sample collection and is primarily focused on preparation to collect specific types of samples. A preliminary step in determining sample collection and analysis methods for a new monitoring project is to examine how sampling was performed under other past or current monitoring efforts in the area or in other locations you may be interested in. As noted in the discussion of trend monitoring (section 2.4.2.4) changes in methods over time can doom the analysis, so it can be very important to align your methods with those used in the past. Unless there is a compelling reason to use different sample collection and analysis methods from those used to generate past data, it may be best to simply use the same methods to increase the likelihood of data compatibility.

3.6.1.1 Documentation and Records

Because field personnel may rotate assignments in a monitoring project, it is critical that field procedures be documented clearly to ensure consistency, both day-to-day and over the long term. Preparation of field manuals and written standard operating procedures (SOPs) will help supplement the basic training that will be required for field personnel. Field personnel should also keep meticulous sample collection records to support and explain the data being collected. These records should include a logbook of calibration and maintenance records for field instruments and notes concerning variations from SOPs, errors, extreme events and field conditions.

3.6.1.2 Preparation for Sampling

Preparation for a sampling trip includes activities such as cleaning, calibrating, and testing field instruments and sampling equipment as well as making certain that all needed supplies and equipment are assembled. The USGS recommends that a formal checklist be filled out in preparation for each sampling trip to make sure that nothing essential is forgotten (Wilde variously dated).

3.6.1.3 Cleaning

Sample containers must be clean to avoid contamination and preserve sample integrity (Wilde 2004). Most water quality variables have specific requirements for the type and composition of sample container and the cleaning process appropriate for that constituent (see section 3.6.3.2 for references to sources of information on analytic methods). Field personnel must ensure that sample containers they take are prepared for use. In the field, most polyethylene sample bottles and those glass sample bottles that are designated for analysis of inorganic constituents should be field rinsed with the same water that will ultimately fill the sample bottle. Specific field rinsing procedures recommended by USGS are described in Table 5-2 of Wilde et al. (2009).

3.6.1.4 Safety

Field personnel are subject to the basic safety policies and regulations of their employer. In addition, field work for water quality monitoring presents special hazards and considerations that should be addressed. Some important safety protocols include:

- Field personnel should not work alone, should have capacity for communication, and should leave contact and itinerary information with their base.
- Pay attention to inclement weather, especially when sampling from boats in open water or sampling in flashy urban streams. Seek shelter or head back to shore if threatening conditions approach.
- When wading to collect samples or make measurements, wear a personal flotation device (PFD) and do not attempt to wade a stream where the depth exceeds 4 ft or where the product of depth (in ft) times velocity (in ft/s) equals or exceeds 8 anywhere in the cross section. This guidance is based on a study that tested the stability of human subjects over a velocity range of 1.2-10 ft/s and a depth range of 1.6-4 ft (Abt et al. 1989).
- When electrofishing (see section 3.6.2.6 and chapter 4), always work in teams of two properly trained technicians and use proper protective equipment.
- Follow standard safety procedures around mechanical equipment and hazardous chemicals.
- Use caution and extra protection when working with water known or suspected to contain high levels of pathogens.

These and other important procedures are documented in detail in chapter 9 of the USGS National Field Manual (Lane and Fay 1997).

3.6.2 Field Procedures

General procedures are discussed below for different types of sampling. The reader is encouraged to consult other resources (e.g., Barbour et al. 1999; USGS variously dated) for more detailed information on specific sampling procedures. A detailed discussion of sample types can be found in section 3.2.

3.6.2.1 Field Measurements

Collection of data on some water quality characteristics must be based on field measurements, rather than samples collected for later analysis in a laboratory. Variables such as water temperature and dissolved oxygen concentration must be measured directly in the waterbody (Figure 3-35). Other properties such as pH, specific conductance, and turbidity can be measured either *in situ* or immediately on the site using a sample taken from the source, depending on the specific instruments involved.

An *in situ* measurement is made by immersing one or more instrument sensors directly into the waterbody. In flowing water, a single sampling point in a well-mixed area is generally used to represent an entire cross-section, often after a preliminary investigation of variability has been made from repeated measurements at points along the cross-section. In lakes or other still water, field measurements may be made at multiple locations and depths, depending on monitoring objectives and the variability of the waterbody. It is important to record the results of individual measurements from the field, not averaged values. Field measurements in ground water generally require purging the monitoring well of standing water before taking measurements so that the measurements accurately represent the properties of the water in the geologic formation at the time of collection. Following purging, field measurements are performed either above ground by pumping water from the well or downhole, using submersible sensors.

Figure 3-35. Measuring dissolved oxygen, specific conductance, pH, and water temperature using a hand-held probe

Detailed procedures for making field measurements are presented in chapter 6 of the USGS National Field Manual (Wilde variously dated).

3.6.2.2 Grab Sampling

There are a variety of devices available to collect grab samples from waterbodies for different purposes (Wilde et al. 2014).

- **Isokinetic depth-integrated samplers** are designed to accumulate a representative water sample continuously and isokinetically (water approaching and entering the sampler intake does not change in velocity) from a vertical section of a stream while transiting the vertical at a uniform rate. Isokinetic samplers may be hand-held or used with cable systems. Such devices are often used for suspended sediment sampling because maintaining constant velocity facilitates the collection of a sample that is representative of all suspended matter moving in the water column. Some examples of isokinetic samplers are shown in Figure 3-36.

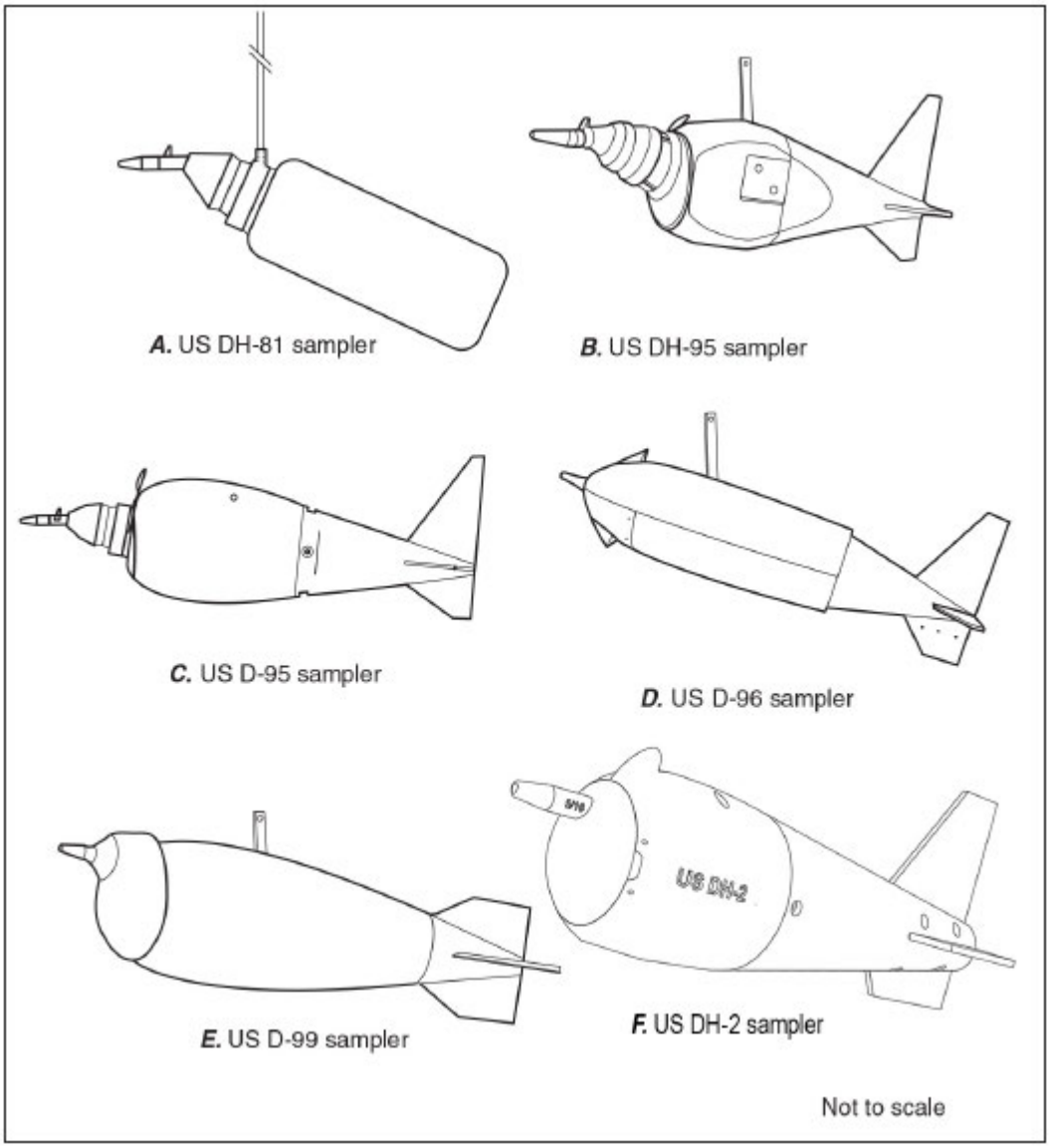
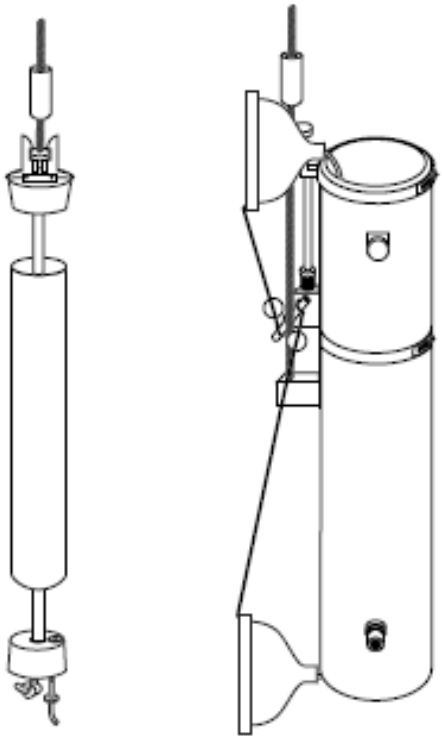


Figure 3-36. Examples of isokinetic depth-integrating samplers (Wilde et al. 2014)

- **Nonisokinetic samplers** are sampling devices in which the sample enters the device at a velocity that differs from ambient. Nonisokinetic samplers include ordinary hand-held open-mouth bottles, weighted bottles on cables, and specialized BOD and volatile organic compound (VOC) samplers for collecting non-aerated samples.
- **Depth-specific samplers** (also called "thief samplers") are used to collect discrete samples from lakes, estuaries and other deep water at a known depth. Common samplers of this type (another form of nonisokinetic sampler) include the Kemmerer and Van Dorn samplers (Figure 3-37).



A. Kemmerer sampler ***B.*** Van Dorn sampler

Figure 3-37. Depth-specific samplers for lake sampling (Wilde et al. 2014)

3.6.2.3 Passive Sampling

Passive samplers are devices to collect unattended grab samples without reliance on external power or electronic activation. They offer the convenience of unattended operation, however in most cases the exact time and circumstance of sampling is unknown unless other data are taken at the same time. Some passive samplers are also limited to collecting samples from the rising limb of the hydrograph, so resulting data may be biased compared to samples collected during the full event. Examples of passive samplers include:

- **Runoff samplers** are used to collect overland flow from urban or rural areas. A first-flush sampler is often a bottle buried so that its mouth is flush with the ground (see Figure 3-30). When the bottle is filled, a check-valve closes, preventing subsequent flow from entering. Another type of runoff sampler/flow splitter collects overland flow and splits off a subsample into a down-slope container. Examples are shown in Figure 3-38.
- **Single-stage samplers** (Figure 3-39) are designed to collect unattended samples for suspended sediment or other constituents from streams during storm events. Multiple units can be mounted above each other to collect samples from different elevations or times as stream stage increases.

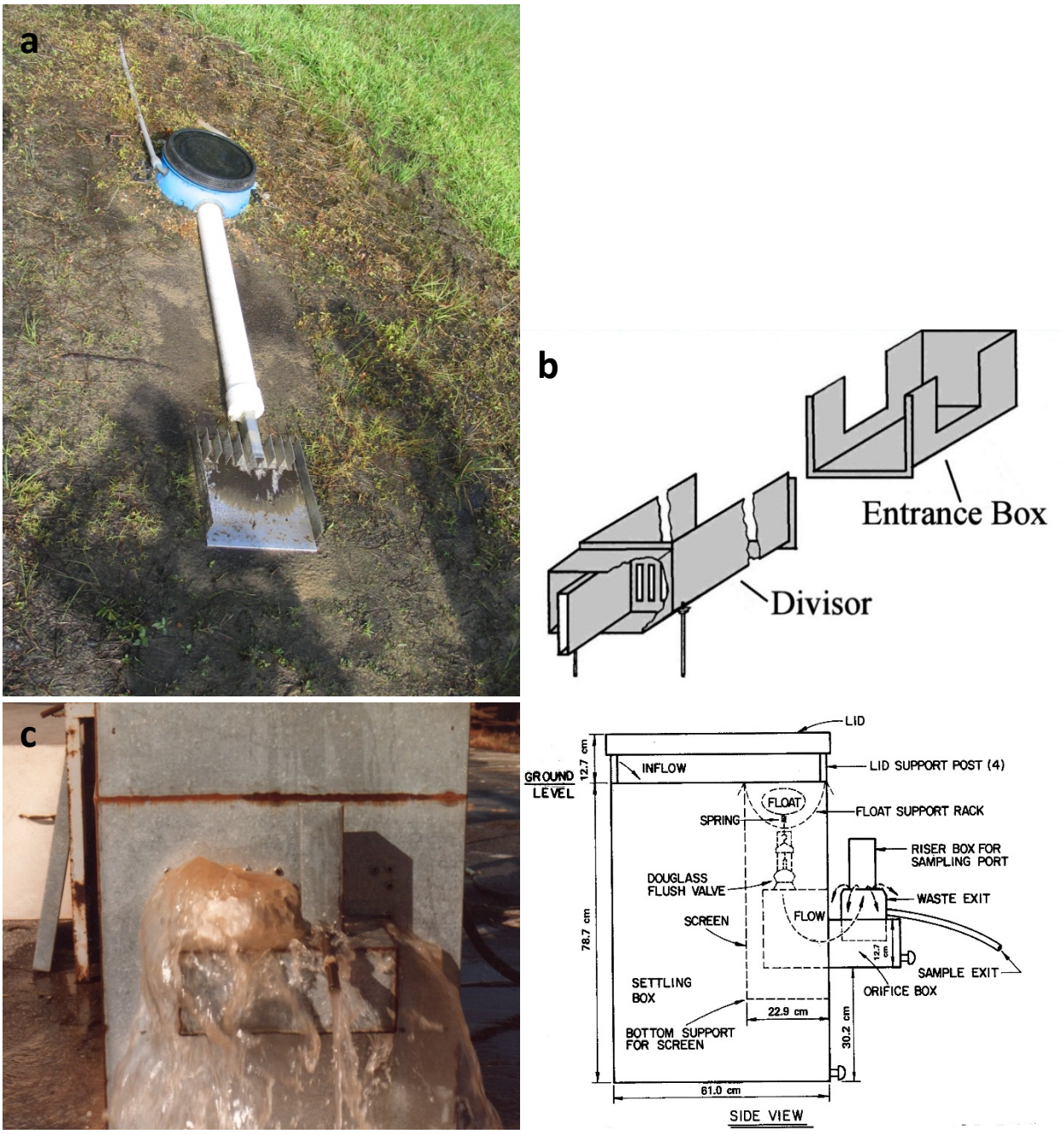


Figure 3-38. Examples of passive samplers. *a,* Passive runoff sampler/flow splitter, University of Georgia, Tifton, GA (photo by D.W. Meals); *b*, Multi-slot divisor (after Brakensiek et al. 1979); *c*, Water and sediment sampler (Dressing et al. 1987, photo by S.A. Dressing).

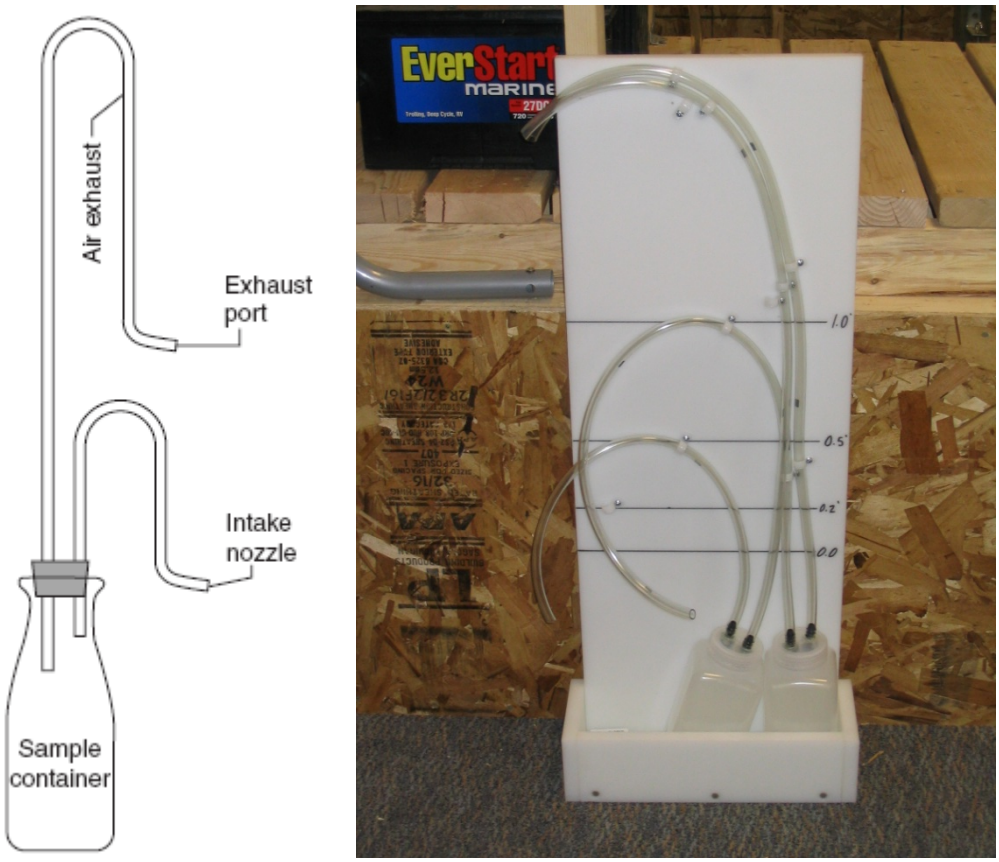


Figure 3-39. Single-stage passive sampler (diagram: Wilde et al. 2014, photo by D.W. Meals)

- **Tipping-bucket samplers** are mechanical devices that capture water flowing from a pipe or other concentrated discharge in one of two pans that tip back and forth on an axis as one pan fills and the other discharges to a large pan. Slots or a funnel can passively convey a sample to a collection bottle; the resulting sample is a flow-proportional composite. A tipping bucket sampler has the additional feature that total discharge can be measured by counting the number of tips with a mechanical counter. An example of design and application of a tipping-bucket system for sampling field runoff and suspended sediment with a pipe collector is given by Kahn and Ong (1997).
- **Coshocton wheel** samplers are rotating wheels driven by the force of water discharging from a pipe or flume (see Figure 3-28). A standing slot collects a sample each time it rotates under the discharge. Coshocton wheels collect a sample volume proportional to the total discharge (usually 1 percent of the discharge) and therefore can provide an estimate of total event discharge.
- **Lysimeters** are devices buried in the ground to sample soil water moving through the vadose zone, the area between the ground surface and water table (Figure 3-40). Lysimeters may be entirely passive ("zero-tension lysimeters") collecting gravitational water in funnels, pans or troughs. Alternatively, tension lysimeters extract a sample of soil water by applying suction through porous plates or cups.

Figure 3-40. Lysimeters before and after installation (photos by R. Traver, Villanova University)

3.6.2.4 Autosampling

Autosamplers generally consist of an intake line submerged in the waterbody or the flow through a pipe or flume, a peristaltic or submersible pump that pumps water to the sampler, one or more bottles to contain collected samplers, and electronic controls to initiate sample collection and record data. Some autosamplers may be refrigerated to preserve samples for extended periods. Some may be designed specifically to fit into storm drains and catch basins. Most operate with either DC or AC power. Examples of autosamplers are shown in Figure 3-41.

Figure 3-41. Examples of portable and refrigerated autosamplers (Hach® 2013b, Teledyne Isco 2013b)

Autosamplers can be set to take time-based samples either continuously, i.e. collect a sample every eight hours, or as initiated by an external trigger such as detection of rainfall or rising stream stage. Some samplers can be set in variable time programs, e.g., to collect samples every 15 minutes during the early part of a storm event, then take hourly samples as the event subsides. When connected to a flow meter, autosamplers can take flow-proportional samples, collecting a subsample for every m^3 that passes the station during a set time period or during a discrete storm event. Flow-proportional sampling may be the most appropriate way of sampling for many NPS pollutants, where high concentrations are associated with high flows and where events that could be missed by timed sampling carry the bulk of the pollutant load (see section 3.2.2.2).

Most autosamplers can collect discrete samples in individual bottles so that a picture of constituent concentration variation across a time period or storm event (i.e., a chemograph) can be plotted and the relationships among time, flow and concentration evaluated. Autosamplers can also combine individual samples into a single larger container to yield a composite sample that represents an extended time period (see section 3.2.2.2). Collecting composite samples can reduce analytical costs by sending a single sample (representing the time period or the storm event) to the laboratory. A flow-proportional sample provides an event mean concentration (EMC) with a single analysis and facilitates load estimation by providing a single EMC result that can be multiplied by the total period or event flow for a load estimate (see section 3.8 and section 7.9).

The flexibility, capacity for self-contained unattended operation, and potential linkage to flow data are major advantages of autosamplers. There are also a few disadvantages with autosamplers. First, autosampler intakes are generally fixed in one position in a waterbody and may therefore not be fully

representative of variability, especially where strong vertical or horizontal gradients exist. Second, the size of the intake line and the velocity achieved by the autosampler pump, as well as the position in the streamflow, may prevent the collection of a representative sample, especially of suspended sediment and particulate-bound pollutants. Third, monitoring for some pollutants like volatile organics or pathogens, may be challenging because of special limitations for materials contacting the sample and requirements for sterilization between sample intake events. Finally, because samples are taken at intervals, regardless of whether an autosampler collects on a time- or flow-based program, the possibility always exists that a transient pulse of a pollutant (e.g., from a spill or first-flush) may pass by unsampled. This of course is also a risk in manual sampling.

Autosamplers must be maintained properly to ensure that sample collection is reliable and performed in accordance with programming instructions. Routine maintenance, sample volume calibration, and probe calibration procedures specified in user manuals should be strictly followed.

3.6.2.5 Benthic Macroinvertebrate Sampling

Sampling of benthic macroinvertebrates from aquatic substrates like stream bottoms and lake beds must consider not only how to physically collect samples but also the diversity of stream habitats that influence the numbers and types of organisms to be sampled. Different types of assemblages of macroinvertebrates inhabit different aquatic habitats (Hawkins et al. 1993). While a monitoring program need not necessarily sample all these habitat types, the habitats sampled should be based on monitoring objectives and on regional stream or lake characteristics. Two distinct types of stream habitats are generally sampled: riffles (shallow areas of fast-moving water, generally with a stony or gravelly bottom) and pools (areas of deeper, slow-flowing water, generally with a softer sediment substrate) (Figure 3-42). In lakes, near-shore areas offer different substrates and habitats from those in deeper lake regions that might lack light, vegetation, and oxygen. Different groups of organisms tend to occupy these habitats, and different approaches for sampling them are required.

The Rapid Bioassessment Protocols (RBPs) recommended by U.S. EPA (Barbour et al. 1999) specify many of the parameters of benthic macroinvertebrate sampling. These issues are discussed in greater detail in chapter 4 of this guidance. In general, benthic macroinvertebrates can be collected actively or passively. In rivers and streams, active collection is often accomplished by disturbing the streambed and capturing the dislodged organisms in a net as the current carries them downstream (Figure 3-43). Kick-seines, D-frame nets, and Surber square-foot samplers are common devices used (Figure 3-44). Regardless of the specific device, it is important to quantify both the area of the streambed disturbed and the time/effort of sampling so that results can be quantified (e.g., organisms/m^2), repeated and compared among different sampling events over time. In lakes, active sampling in shallow areas can be done by similar methods. Grab samplers such as the petite ponar (Figure 3-44) or larger dredges are used for taking sediment samples from hard bottoms such as sand, gravel).

Passive sampling for benthic macroinvertebrates often uses artificial substrates like the Hester-Dendy plate sampler or rock baskets (Figure 3-44) that are anchored in the waterbody. Organisms colonize the devices and then the devices are retrieved to collect and enumerate the organisms.

All of these techniques have advantages and disadvantages that are discussed in chapter 4.

Figure 3-42. Preparing to take samples in a low-gradient stream

Figure 3-43. Using a D-frame net to sample a gravel bottom stream for benthic macroinvertebrates

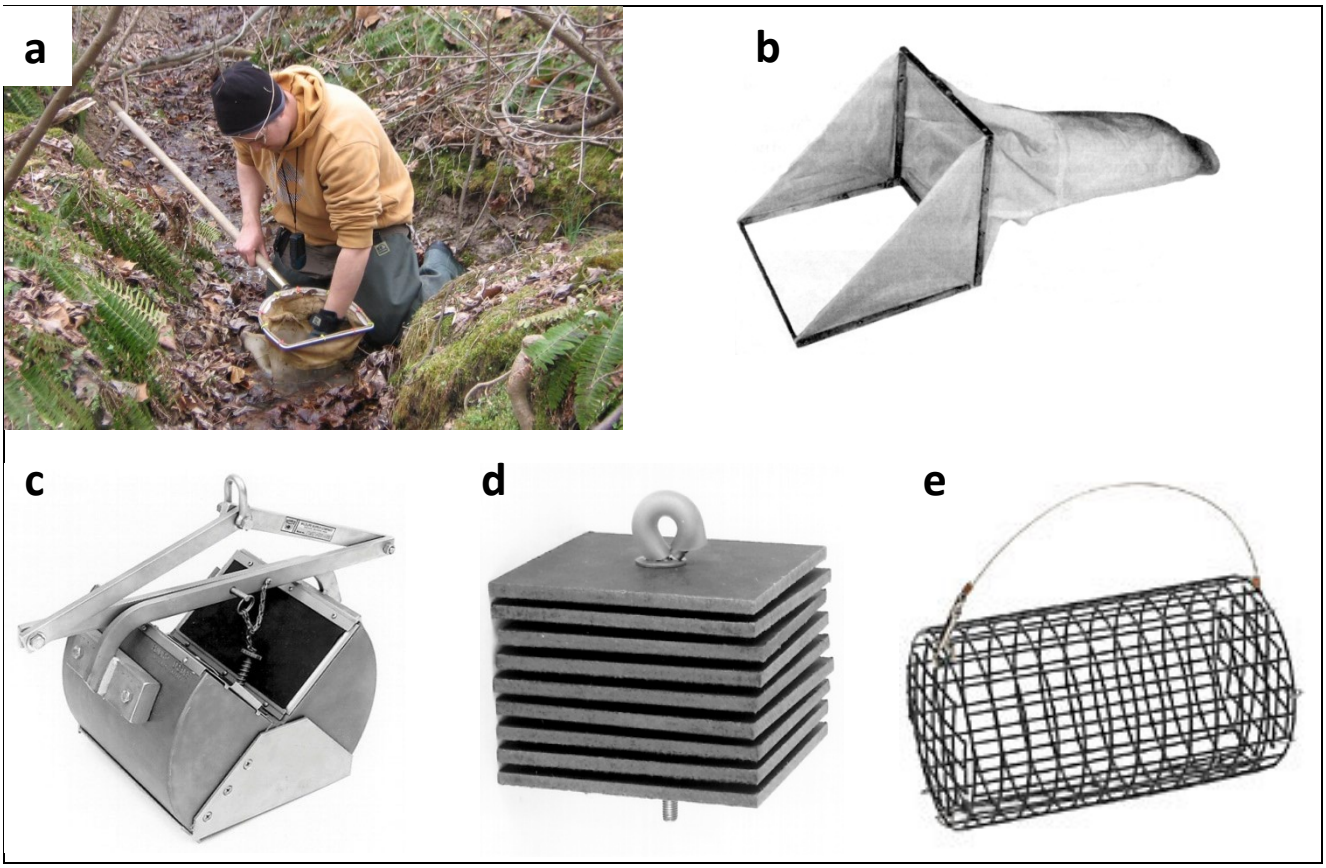


Figure 3-44. Sampling devices for biological and habitat variables. *a,* D-frame net; *b*, Surber sampler (Rickly 2016); *c*, Ponar dredge (Rickly 2016); *d*, Hester-Dendy artificial substrate (Rickly 2016); *e*, Rock basket artificial substrate (Ben Meadows 2016).

3.6.2.6 Fish Sampling

As with benthic macroinvertebrates, distinct fish assemblages are found in different habitat types. For fish, characteristics like water temperature, flow velocity, dissolved oxygen levels, cover and shade, in addition to substrate type, are important habitat characteristics. In general, biomonitoring efforts should sample fish habitats based on project objectives and resource characteristics. Major habitat types like riffles, pools and runs (stream reaches between riffles and pools) should normally be sampled. Habitats and the size of sampling areas should be consistent between sampling events to allow long-term comparisons.

Fish are most commonly sampled by electrofishing, where a portable generator system introduces an electric current into the water, temporarily stunning fish within a certain range (Figure 3-45). In practice, the ends of a sampling reach (approximately 30 m in length) are closed off with nets and a sampling crew walks through the reach. One person runs the shocker, while the others retrieve stunned fish into buckets. When collection is complete, the fish are counted and identified (usually to species), then returned to the stream (Figure 3-46). The process may be repeated at several different sites of similar habitat to ensure a representative sampling has been achieved. Other approaches to fish sampling include use of seines, gill nets, traps, or underwater observation. For a discussion of the advantages and limitations of different fish sampling gear, see Klemm et al. (1992). Ohio EPA (OEPA 1987) discusses electrofishing techniques for bioassessment.

Figure 3-45. Backpack electrofishing (USEPA)

Figure 3-46. Field processing of fish sample: taxonomic identification and data recording

3.6.2.7 Aquatic Plant Sampling

Aquatic plants sampled for water quality monitoring include algae (small free-floating plants), periphyton (the community of algae, microbes, and detritus attached to submerged surfaces), and macrophytes (large, plants rooted in aquatic sediments). Many of these plants are good indicators of nutrient enrichment and ecosystem condition. Algae are usually evaluated in lakes or other bodies of standing water and are sampled using a plankton net towed through the water column (Figure 3-47). Collected organisms are identified and counted under a microscope. As a surrogate for algal biomass, chemical analysis of a water sample for chlorophyll *a* may be performed. Periphyton biomass is usually measured in streams, either by scraping known areas of rock surfaces or by use of artificial substrates (typically glass microscope slides) placed in the stream and retrieved after a specified period. Aquatic macrophytes, often monitored in near-shore areas of lakes or in large rivers, may be surveyed to assess species composition, quantified in small plots by counting individual plants or harvesting vegetation, or mapped by remote sensing to document areal extent of growth.

Figure 3-47. Plankton nets (NOAA 2014)

3.6.2.8 Bacteria/Pathogen Sampling

Collection of water samples for monitoring indicator bacteria, pathogens, or other microorganisms is usually conducted by grab sampling. Samples for fecal coliform and *E. coli* bacteria analysis typically require small volumes (e.g., 100 milliliter [ml]). Samples for detection and enumeration of protozoan pathogens like *Giardia* and *Cryptosporidium* may require up to 20 Liter (L) of sample. Sterile sample containers such as autoclaved polyethylene containers or pre-sterilized single-use bags or bottles are required. Sample collection should be done by clean technique, with samples allowed to contact only sterile surfaces; field personnel should wear gloves when collecting grab samples, both to protect themselves from water-borne pathogens and to prevent sample contamination. Samples for bacteria

and/or pathogens might require more rapid delivery to the laboratory than samples from physical and chemical analysis (see section 3.6.3).

3.6.2.9 Habitat Sampling

Assessment of aquatic habitat may be essential to interpretation of data collected from monitoring of benthic invertebrates and fish. Habitat characteristics might also be an important response variable to land treatment or stream restoration efforts. Habitat quality may be measured in three dimensions: habitat structure, flow regime, and energy source. Habitat structure includes physical characteristics of stream environment such as channel morphology, gradient, instream cover (boulders and woody debris), substrate types, riparian condition, and bank stability. Flow regime is defined by velocity and volume of water moving through a stream, both on the average and during extreme events (wet or dry). Energy enters stream systems through nutrients from runoff or ground water, as leaves and other debris falling into streams, or from photosynthesis by aquatic plants and algae.

Some important metrics of habitat sampling were shown in Table 3-7 in section 3.1.4. Many habitat characteristics are quantified by direct measurement in representative stream reaches, e.g., by surveying, substrate sampling, and soil/geophysical measurements. Sets of habitat measurements are often incorporated into indices that facilitate comparison between sites and between sampling times. For example, the Qualitative Habitat Evaluation Index (QHEI) used by Ohio EPA (Rankin 1989) includes measurements of:

- **Substrate:** type and quality
- **Instream cover:** type and amount
- **Channel morphology:** sinuosity, development, channelization, stability
- **Riparian zone:** width, quality, bank erosion
- **Pool quality:** maximum depth, current, morphology
- **Riffle quality:** depth, substrate stability, substrate embeddedness
- **Map gradient**

Habitat assessment is discussed further in chapter 4 of this guidance. The reader is referred to additional resources for more information on habitat sampling:

- Rapid Bioassessment Protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish (Barbour et al. 1999)
- The Qualitative Habitat Evaluation Index (QHEI): rationale, methods, and application (Rankin 1989)
- Methods for assessing habitat in flowing waters: using the Qualitative Habitat Evaluation Index (QHEI) (OEPA 2006.).

3.6.2.10 Specialized Sampling

Specialized sampling techniques may be required for unusual or emerging pollutants. For example, microbial source tracking analyzes DNA to attribute indicator bacteria to specific host sources (USEPA 2011b, Meals et al. 2013). This method requires water sampling and might also involve collection of fecal material from human and animal sources in the watershed.

Urban stormwater monitoring may test for optical brighteners (fluorescent whitening agents added to laundry detergent) in stormwater as indicators of wastewater or septic effluent contamination. Because

these chemicals are absorbed by fabric, cotton pads are deployed in streams for several days, then collected and tested for fluorescence with a UV source (Gilpin et al. 2002).

Sentinel chambers, dialysis membrane diffusion samplers, polar organic chemical integrative samplers (POCIS), and other passive sampling devices have been used to passively sample low-concentration pollutants like volatile organic compounds, estrogen analogs, endocrine disruptors, and other emerging pollutants in a variety of settings (Vrana et al. 2005, Liscio et al. 2009, Kuster et al. 2010).

3.6.3 From Field to Laboratory

There are several important steps to consider between sample collection and analysis including sample processing, sample preservation and transport, sample custody tracking, and performance audits. Quality assurance and quality control procedures are described in detail in chapter 8.

3.6.3.1 Sample Processing

Sample processing refers to the measures taken to prepare and preserve a water sample at or after collection, and before it is delivered to the laboratory for analysis. The goals of sample processing are to prepare samples for appropriate analysis (e.g., dissolved vs. TP), prevent contamination and cross-contamination, and preserve sample integrity until analysis. The USGS National Field Manual includes detailed sample processing procedures for many specific analytes, and recommends the following order of sample processing: organic fraction, organic C, inorganic constituents, nutrients, radiochemicals, isotopes, and then microorganisms (Wilde et al. 2009).

Samples requiring filtration (e.g., dissolved P, dissolved organic C) must be filtered during or immediately after collection (Wilde et al. 2009). Surface water samples may be composited or subsampled in the field using an appropriate device, such as a churn or cone splitter (Figure 3-48). Ground water samples are not composited but are pumped either directly through a splitter or through a filtration assembly into sample bottles unless a bailer or other downhole sampler is used to collect the sample.

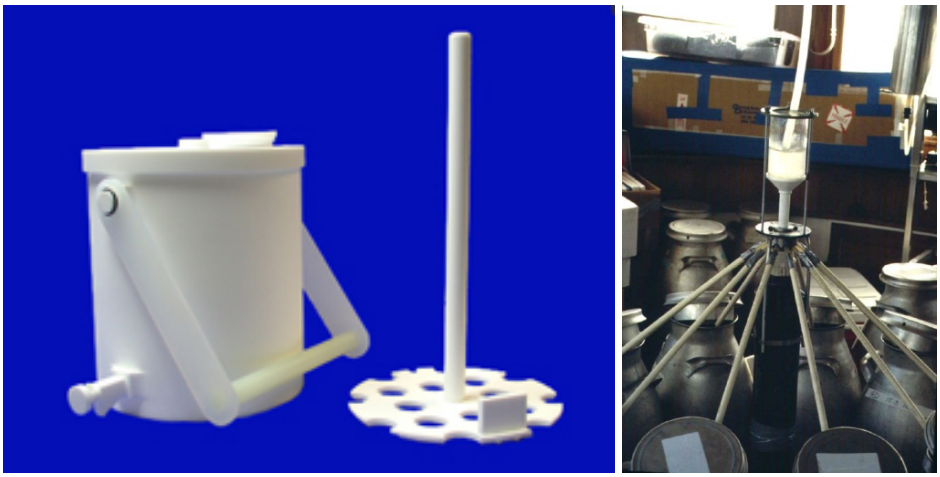


Figure 3-48. Churn and cone splitters (FISP 2014)

3.6.3.2 Sample Preservation and Transport

Water samples to be analyzed for most water quality variables have specified permissible holding time and holding conditions that determine the length of time a sample can be held between collection and analysis without significantly affecting the analytical results. Maximum holding times and storage

conditions have been established by the EPA (40 CFR 136.3, USEPA 2008b) and are shown in Table 3-12. Storage and preservation for most analytes involve cooling to below 6 ºC; chemical preservatives such as nitric acid (HNO_3) or sulfuric acid (H_2SO_4) may also be used, depending on the analyte (Wilde et al. 2009).

Samples should be packaged and transported to the laboratory for analysis as soon as possible. The shorter the time between sample collection and analysis, the more reliable the analytical results will be. If samples must be shipped to a laboratory, check to insure that sample containers are sealed, labeled, and packed to prevent breakage. It is necessary to follow receiving laboratory protocols for labeling, documenting, and packaging samples.

Table 3-12. Required containers, preservation techniques, and holding times

Parameter number/name	Container [1]	Preservation [2,3]	Maximum holding time [4]
Table IA—Bacterial Tests:			
1-5. Coliform, total, fecal, and ***E. coli***	PA, G	Cool, <10 °C, 0.0008% $Na_2S_2O_3$[5]	8 hours.[22,23]
6. Fecal streptococci	PA, G	Cool, <10 °C, 0.0008% $Na_2S_2O_3$[5]	8 hours.[22]
7. Enterococci	PA, G	Cool, <10 °C, 0.0008% $Na_2S_2O_3$[5]	8 hours.[22]
8.*Salmonella*	PA, G	Cool, <10 °C, 0.0008% $Na_2S_2O_3$[5]	8 hours.[22]
Table IA—Aquatic Toxicity Tests:			
9-12. Toxicity, acute and chronic	P, FP, G	Cool, ≤6 °C [16]	36 hours.
Table IB—Inorganic Tests:			
1. Acidity	P, FP, G	Cool, ≤6 °C [18]	14 days.
2. Alkalinity	P, FP, G	Cool, ≤6 °C [18]	14 days.
4. Ammonia	P, FP, G	Cool, ≤6 °C [18], $H_2SO_4$to pH <2	28 days.
9. Biochemical oxygen demand	P, FP, G	Cool, ≤6 °C [18]	48 hours.
10. Boron	P, FP, or Quartz	$HNO_3$to pH <2	6 months.
11. Bromide	P, FP, G	None required	28 days.
14. Biochemical oxygen demand, carbonaceous	P, FP G	Cool, ≤6 °C [18]	48 hours.
15. Chemical oxygen demand	P, FP, G	Cool, ≤6 °C [18], $H_2SO_4$to pH <2	28 days.
16. Chloride	P, FP, G	None required	28 days.
17. Chlorine, total residual	P, G	None required	Analyze within 15 minutes.
21. Color	P, FP, G	Cool, ≤6 °C [18]	48 hours.
23-24. Cyanide, total or available (or CATC) and free	P, FP, G	Cool, ≤6 °C [18], NaOH to pH >10 [5 6], reducing agent if oxidizer present	14 days.
25. Fluoride	P	None required	28 days.
27. Hardness	P, FP, G	HNO_3or $H_2SO_4$to pH <2	6 months.
28. Hydrogen ion (pH)	P, FP, G	None required	Analyze within 15 minutes.
31, 43. Kjeldahl and organic N	P, FP, G	Cool, ≤6 °C [18], $H_2SO_4$to pH <2	28 days.
Table IB—Metals: [7]			
18. Chromium VI	P, FP, G	Cool, ≤6 °C [18], pH = 9.3-9.7 [20]	28 days.
35. Mercury (CVAA)	P, FP, G	$HNO_3$to pH <2	28 days.
35. Mercury (CVAFS)	FP, G; and FP-lined cap [17]	5 ml/L 12N HCl or 5 ml/L BrCl [17]	90 days.[17]

Parameter number/name	Container [1]	Preservation [2,3]	Maximum holding time [4]
3, 5-8, 12, 13, 19, 20, 22, 26, 29, 30, 32-34, 36, 37, 45, 47, 51, 52, 58-60, 62, 63, 70-72, 74, 75. Metals, except boron, chromium VI, and mercury	P, FP, G	$HNO_3$to pH <2, or at least 24 hours prior to analysis [19]	6 months.
38. Nitrate	P, FP, G	Cool, ≤6 °C [18]	48 hours.
39. Nitrate-nitrite	P, FP, G	Cool, ≤6 °C [18], $H_2SO_4$to pH <2	28 days.
40. Nitrite	P, FP, G	Cool, ≤6 °C [18]	48 hours.
41. Oil and grease	G	Cool to ≤6 °C [18], HCl or $H_2SO_4$to pH <2	28 days.
42. Organic Carbon	P, FP, G	Cool to ≤6 °C [18], HCl, H_2SO_4, or $H_3PO_4$to pH <2	28 days.
44. Orthophosphate	P, FP, G	Cool, to ≤6 °C [18,24]	Filter within 15 minutes; Analyze within 48 hours.
46. Oxygen, Dissolved Probe	G, Bottle and top	None required	Analyze within 15 minutes.
47. Winkler	G, Bottle and top	Fix on site and store in dark	8 hours.
48. Phenols	G	Cool, ≤6 °C [18], $H_2SO_4$to pH <2	28 days.
49. Phosphorous (elemental)	G	Cool, ≤6 °C [18]	48 hours.
50. Phosphorous, total	P, FP, G	Cool, ≤6 °C [18], $H_2SO_4$to pH <2	28 days.
53. Residue, total	P, FP, G	Cool, ≤6 °C [18]	7 days.
54. Residue, Filterable	P, FP, G	Cool, ≤6 °C [18]	7 days.
55. Residue, Nonfilterable (TSS)	P, FP, G	Cool, ≤6 °C [18]	7 days.
56. Residue, Settleable	P, FP, G	Cool, ≤6 °C [18]	48 hours.
57. Residue, Volatile	P, FP, G	Cool, ≤6 °C [18]	7 days.
61. Silica	P or Quartz	Cool, ≤6 °C [18]	28 days.
64. Specific conductance	P, FP, G	Cool, ≤6 °C [18]	28 days.
65. Sulfate	P, FP, G	Cool, ≤6 °C [18]	28 days.
66. Sulfide	P, FP, G	Cool, ≤6 °C [18], add zinc acetate plus sodium hydroxide to pH >9	7 days.
67. Sulfite	P, FP, G	None required	Analyze within 15 minutes.
68. Surfactants	P, FP, G	Cool, ≤6 °C [18]	48 hours.
69. Temperature	P, FP, G	None required	Analyze.
73. Turbidity	P, FP, G	Cool, ≤6 °C [18]	48 hours.
Table IC—Organic Tests: [8]			
13, 18-20, 22, 24-28, 34-37, 39-43, 45-47, 56, 76, 104, 105, 108-111, 113. Purgeable Halocarbons	G, FP-lined septum	*Cool, ≤6 °C [18], 0.008% $Na_2S_2O_3$[5]*	14 days.
6, 57, 106. Purgeable aromatic hydrocarbons	G, FP-lined septum	*Cool, ≤6 °C [18], 0.008% $Na_2S_2O_3$[5], HCl to pH 2 [9]*	14 days.[9]
3, 4. Acrolein and acrylonitrile	G, FP-lined septum	*Cool, ≤6 °C [18], 0.008% $Na_2S_2O_3$, pH to 4-5[10]*	14 days.[10]
23, 30, 44, 49, 53, 77, 80, 81, 98, 100, 112. Phenols [11]	G, FP-lined cap	*Cool, ≤6 °C [18], 0.008% $Na_2S_2O_3$*	7 days until extraction, 40 days after extraction.
7, 38. Benzidines [11 12]	G, FP-lined cap	*Cool, ≤6 °C [18], 0.008% $Na_2S_2O_3$[5]*	7 days until extraction.[13]

Parameter number/name	Container [1]	Preservation [2,3]	Maximum holding time [4]
14, 17, 48, 50-52. Phthalate esters [11]	G, FP-lined cap	*Cool, ≤6 °C [18]*	7 days until extraction, 40 days after extraction.
82-84. Nitrosamines [11 14]	G, FP-lined cap	*Cool, ≤6 °C[18], store in dark, 0.008% $Na_2S_2O_3$[5]*	7 days until extraction, 40 days after extraction.
88-94. PCBs [11]	G, FP-lined cap	*Cool, ≤6 °C [18]*	1 year until extraction, 1 year after extraction.
54, 55, 75, 79. Nitroaromatics and isophorone [11]	G, FP-lined cap	*Cool, ≤6 °C [18], store in dark, 0.008% $Na_2S_2O_3$[5]*	7 days until extraction, 40 days after extraction.
1, 2, 5, 8-12, 32, 33, 58, 59, 74, 78, 99, 101. Polynuclear aromatic hydrocarbons [11]	G, FP-lined cap	*Cool, ≤6 °C [18], store in dark, 0.008% $Na_2S_2O_3$[5]*	7 days until extraction, 40 days after extraction.
15, 16, 21, 31, 87. Haloethers [11]	G, FP-lined cap	*Cool, ≤6 °C [18], 0.008% $Na_2S_2O_3$[5]*	7 days until extraction, 40 days after extraction.
29, 35-37, 63-65, 107. Chlorinated hydrocarbons [11]	G, FP-lined cap	*Cool, ≤6 °C [18]*	7 days until extraction, 40 days after extraction.
60-62, 66-72, 85, 86, 95-97, 102, 103. CDDs/CDFs [11]			
Aqueous Samples: Field and Lab Preservation	G	*Cool, ≤6 °C [18], 0.008% $Na_2S_2O_3$[5], pH <9*	1 year.
Solids and Mixed-Phase Samples: Field Preservation	G	*Cool, ≤6 °C [18]*	7 days.
Tissue Samples: Field Preservation	G	*Cool, ≤6 °C [18]*	24 hours.
Solids, Mixed-Phase, and Tissue Samples: Lab Preservation	G	*Freeze, ≤ −10 °C*	1 year.
114-118. Alkylated phenols	G	*Cool, <6 °C, $H_2SO_4$to pH <2*	28 days until extraction, 40 days after extraction.
119. Adsorbable Organic Halides (AOX)	G	*Cool, <6 °C, 0.008% $Na_2S_2O_3$$HNO_3$to pH <2*	Hold***at least***3 days, but not more than 6 months.
120. Chlorinated Phenolics		*Cool, <6 °C, 0.008% $Na_2S_2O_3$$H_2SO_4$to pH <2*	30 days until acetylation, 30 days after acetylation.
Table ID—Pesticides Tests:			
1-70. Pesticides [11]	G, FP-lined cap	*Cool, ≤6 °C [18], pH 5-9-[15]*	7 days until extraction, 40 days after extraction.
Table IE—Radiological Tests:			
1-5. Alpha, beta, and radium	P, FP, G	*$HNO_3$to pH <2*	6 months.
Table IH—Bacterial Tests:			
1.***E. coli***	PA, G	*Cool, <10 °C, 0.0008% $Na_2S_2O_3$[5]*	8 hours.[22]
2. Enterococci	PA, G	*Cool, <10 °C, 0.0008% $Na_2S_2O_3$[5]*	8 hours.[22]
Table IH—Protozoan Tests:			

Parameter number/name	Container [1]	Preservation [2,3]	Maximum holding time [4]
8.***Cryptosporidium***	LDPE; field filtration	*1-10 °C*	96 hours.[21]
9.***Giardia***	LDPE; field filtration	*1-10 °C*	96 hours.[21]

[1] "P" is for polyethylene; "FP" is fluoropolymer (polytetrafluoroethylene (PTFE); Teflon®), or other fluoropolymer, unless stated otherwise in this Table II; "G" is glass; "PA" is any plastic that is made of a sterilizable material (polypropylene or other autoclavable plastic); "LDPE" is low density polyethylene.

[2] Except where noted in this Table II and the method for the parameter, preserve each grab sample within 15 minutes of collection. For a composite sample collected with an automated sample (e.g., using a 24-hour composite sample; see 40 CFR 122.21(g)(7)(i) or 40 CFR Part 403, Appendix E), refrigerate the sample at ≤ 6 °C during collection unless specified otherwise in this Table II or in the method(s). For a composite sample to be split into separate aliquots for preservation and/or analysis, maintain the sample at ≤ 6 °C, unless specified otherwise in this Table II or in the method(s), until collection, splitting, and preservation is completed. Add the preservative to the sample container prior to sample collection when the preservative will not compromise the integrity of a grab sample, a composite sample, or aliquot split from a composite sample within 15 minutes of collection. If a composite measurement is required but a composite sample would compromise sample integrity, individual grab samples must be collected at prescribed time intervals (e.g., 4 samples over the course of a day, at 6-hour intervals). Grab samples must be analyzed separately and the concentrations averaged. Alternatively, grab samples may be collected in the field and composited in the laboratory if the compositing procedure produces results equivalent to results produced by arithmetic averaging of results of analysis of individual grab samples. For examples of laboratory compositing procedures, see EPA Method 1664 Rev. A (oil and grease) and the procedures at 40 CFR 141.34(f)(14)(iv) and (v) (volatile organics).

[3] When any sample is to be shipped by common carrier or sent via the U.S. Postal Service, it must comply with the Department of Transportation Hazardous Materials Regulations (49 CFR part 172). The person offering such material for transportation is responsible for ensuring such compliance. For the preservation requirement of Table II, the Office of Hazardous Materials, Materials Transportation Bureau, Department of Transportation has determined that the Hazardous Materials Regulations do not apply to the following materials: Hydrochloric acid (HCl) in water solutions at concentrations of 0.04% by weight or less (pH about 1.96 or greater; Nitric acid (HNO3) in water solutions at concentrations of 0.15% by weight or less (pH about 1.62 or greater); Sulfuric acid (H2SO4) in water solutions at concentrations of 0.35% by weight or less (pH about 1.15 or greater); and Sodium hydroxide (NaOH) in water solutions at concentrations of 0.080% by weight or less (pH about 12.30 or less).

[4] Samples should be analyzed as soon as possible after collection. The times listed are the maximum times that samples may be held before the start of analysis and still be considered valid. Samples may be held for longer periods only if the permittee or monitoring laboratory has data on file to show that, for the specific types of samples under study, the analytes are stable for the longer time, and has received a variance from the Regional Administrator under Sec. 136.3(e). For a grab sample, the holding time begins at the time of collection. For a composite sample collected with an automated sampler (e.g., using a 24-hour composite sampler; see 40 CFR 122.21(g)(7)(i) or 40 CFR part 403, Appendix E), the holding time begins at the time of the end of collection of the composite sample. For a set of grab samples composited in the field or laboratory, the holding time begins at the time of collection of the last grab sample in the set. Some samples may not be stable for the maximum time period given in the table. A permittee or monitoring laboratory is obligated to hold the sample for a shorter time if it knows that a shorter time is necessary to maintain sample stability. See 136.3(e) for details. The date and time of collection of an individual grab sample is the date and time at which the sample is collected. For a set of grab samples to be composited, and that are all collected on the same calendar date, the date of collection is the date on which the samples are collected. For a set of grab samples to be composited, and that are collected across two calendar dates, the date of collection is the dates of the two days; e.g., November 14-15. For a composite sample collected automatically on a given date, the date of collection is the date on which the sample is collected. For a composite sample collected automatically, and that is collected across two calendar dates, the date of collection is the dates of the two days; e.g., November 14-15. For static-renewal toxicity tests, each grab or composite sample may also be used to prepare test solutions for renewal at 24 h, 48 h, and/or 72 h after first use, if stored at 0-6 °C, with minimum head space.

[5] ASTM D7365-09a specifies treatment options for samples containing oxidants (e.g.,chlorine). Also, Section 9060A of Standard Methods for the Examination of Water and Wastewater (20th and 21st editions) addresses dechlorination procedures.

[6] Sampling, preservation and mitigating interferences in water samples for analysis of cyanide are described in ASTM D7365-09a. There may be interferences that are not mitigated by the analytical test methods or D7365-09a. Any technique for removal or suppression of interference may be employed, provided the laboratory demonstrates that it more accurately measures cyanide through quality control measures described in the analytical test method. Any removal or suppression technique not described in D7365-09a or the analytical test method must be documented along with supporting data.

[7] For dissolved metals, filter grab samples within 15 minutes of collection and before adding preservatives. For a composite sample collected with an automated sampler (e.g., using a 24-hour composite sampler; see 40 CFR 122.21(g)(7)(i) or 40 CFR Part 403, Appendix E), filter the sample within 15 minutes after completion of collection and before adding preservatives. If it is known or suspected that dissolved sample integrity will be compromised during collection of a composite sample collected automatically over time (e.g., by interchange of a metal between dissolved and suspended forms), collect and filter grab samples to be composited (footnote 2) in place of a composite sample collected automatically.

[8] Guidance applies to samples to be analyzed by GC, LC, or GC/MS for specific compounds.

[9] If the sample is not adjusted to pH 2, then the sample must be analyzed within seven days of sampling.

[10] The pH adjustment is not required if acrolein will not be measured. Samples for acrolein receiving no pH adjustment must be analyzed within 3 days of sampling.

[11] When the extractable analytes of concern fall within a single chemical category, the specified preservative and maximum holding times should be observed for optimum safeguard of sample integrity (i.e.,use all necessary preservatives and hold for the shortest time listed). When the analytes of concern fall within two or more chemical categories, the sample may be preserved by cooling to ≤ 6 °C, reducing residual chlorine with 0.008% sodium

thiosulfate, storing in the dark, and adjusting the pH to 6-9; samples preserved in this manner may be held for seven days before extraction and for forty days after extraction. Exceptions to this optional preservation and holding time procedure are noted in footnote 5 (regarding the requirement for thiosulfate reduction), and footnotes 12, 13 (regarding the analysis of benzidine).

[12] If 1,2-diphenylhydrazine is likely to be present, adjust the pH of the sample to 4.0 ± 0.2 to prevent rearrangement to benzidine.

[13] Extracts may be stored up to 30 days at < 0 °C.

[14] For the analysis of diphenylnitrosamine, add 0.008% Na2S2O3and adjust pH to 7-10 with NaOH within 24 hours of sampling.

[15] The pH adjustment may be performed upon receipt at the laboratory and may be omitted if the samples are extracted within 72 hours of collection. For the analysis of aldrin, add 0.008% Na2S2O3.

[16] Place sufficient ice with the samples in the shipping container to ensure that ice is still present when the samples arrive at the laboratory. However, even if ice is present when the samples arrive, immediately measure the temperature of the samples and confirm that the preservation temperature maximum has not been exceeded. In the isolated cases where it can be documented that this holding temperature cannot be met, the permittee can be given the option of on-site testing or can request a variance. The request for a variance should include supportive data which show that the toxicity of the effluent samples is not reduced because of the increased holding temperature. Aqueous samples must not be frozen. Hand-delivered samples used on the day of collection do not need to be cooled to 0 to 6 °C prior to test initiation.

[17] Samples collected for the determination of trace level mercury (<100 ng/L) using EPA Method 1631 must be collected in tightly-capped fluoropolymer or glass bottles and preserved with BrCl or HCl solution within 48 hours of sample collection. The time to preservation may be extended to 28 days if a sample is oxidized in the sample bottle. A sample collected for dissolved trace level mercury should be filtered in the laboratory within 24 hours of the time of collection. However, if circumstances preclude overnight shipment, the sample should be filtered in a designated clean area in the field in accordance with procedures given in Method 1669. If sample integrity will not be maintained by shipment to and filtration in the laboratory, the sample must be filtered in a designated clean area in the field within the time period necessary to maintain sample integrity. A sample that has been collected for determination of total or dissolved trace level mercury must be analyzed within 90 days of sample collection.

[18] Aqueous samples must be preserved at ≤ 6 °C, and should not be frozen unless data demonstrating that sample freezing does not adversely impact sample integrity is maintained on file and accepted as valid by the regulatory authority. Also, for purposes of NPDES monitoring, the specification of "≤ °C" is used in place of the "4 °C" and "< 4 °C" sample temperature requirements listed in some methods. It is not necessary to measure the sample temperature to three significant figures (1/100th of 1 degree); rather, three significant figures are specified so that rounding down to 6 °C may not be used to meet the ≤6 °C requirement. The preservation temperature does not apply to samples that are analyzed immediately (less than 15 minutes).

[19] An aqueous sample may be collected and shipped without acid preservation. However, acid must be added at least 24 hours before analysis to dissolve any metals that adsorb to the container walls. If the sample must be analyzed within 24 hours of collection, add the acid immediately (see footnote 2). Soil and sediment samples do not need to be preserved with acid. The allowances in this footnote supersede the preservation and holding time requirements in the approved metals methods.

[20] To achieve the 28-day holding time, use the ammonium sulfate buffer solution specified in EPA Method 218.6. The allowance in this footnote supersedes preservation and holding time requirements in the approved hexavalent chromium methods, unless this supersession would compromise the measurement, in which case requirements in the method must be followed.

[21] Holding time is calculated from time of sample collection to elution for samples shipped to the laboratory in bulk and calculated from the time of sample filtration to elution for samples filtered in the field.

[22] Sample analysis should begin as soon as possible after receipt; sample incubation must be started no later than 8 hours from time of collection.

[23] For fecal coliform samples for sewage sludge (biosolids) only, the holding time is extended to 24 hours for the following sample types using either EPA Method 1680 (LTB-EC) or 1681 (A-1): Class A composted, Class B aerobically digested, and Class B anaerobically digested.

[24] The immediate filtration requirement in orthophosphate measurement is to assess the dissolved or bio-available form of orthophosphorus (i.e.,that which passes through a 0.45-micron filter), hence the requirement to filter the sample immediately upon collection (i.e.,within 15 minutes of collection).

[38 FR 28758, Oct. 16, 1973]

Source: Electronic Code of Federal Regulations, U.S. Government Printing Office (http://www.ecfr.gov)
Title 40: Protection of Environment
PART 136—GUIDELINES ESTABLISHING TEST PROCEDURES FOR THE ANALYSIS OF POLLUTANTS , § 136.3 Identification of test procedures.
(Accessed January 29, 2016).

3.6.3.3 Sample Custody

The location and status of collected samples must be tracked at all points between the source waterbody and the final data report (see chapter 8). The purposes of tracking sample custody are to prevent loss of samples and/or data, document the conditions under which the samples were held between collection and analysis, and preserve sample and data security and integrity. The principal goal is to be able to track each individual analytical result back through all the steps between collection and analysis should any questions arise concerning analytical results. Records of sample custody are important in all monitoring programs, but are especially critical where data may be used for regulatory or litigation purposes.

Sample custody starts with a consistent numbering and labeling system that uniquely identifies each sample with respect to source, monitoring program, date and time of collection, responsible person(s), and

desired analysis. Custody is usually tracked through forms and other records that are signed and dated by each individual in the chain. For example, in addition to field logs and notes, field personnel will generally fill out a form upon delivery of samples to the laboratory documenting sample identification numbers, program name, date and time of collection, date and time of delivery, and name of delivery person. Laboratory staff will incorporate sample identification numbers into their own custody and data tracking system.

3.6.3.4 Performance Audits

Regular field operations performance audits should be part of the overall quality assurance/quality control process embodied in the QAPP (see chapter 8). These performance audits might include actions such as:

- **Sample container and equipment blanks:** distilled/deionized water is processed through sampling equipment and sample containers to rule out contamination.
- **Trip blanks:** distilled/deionized water is transported from the laboratory through the field sampling process to document any potential contamination during travel and transport.
- **Field duplicates:** two grab samples are collected in quick succession to assess repeatability of sampling.
- **Field splits:** a collected sample is split into two subsamples to assess analytical performance by the laboratory or to make comparisons between labs.

3.6.4 Laboratory Considerations

Water quality samples collected from field sites are generally analyzed in a laboratory. While field test kits are widely available and commonly used in volunteer/citizen monitoring, the accuracy and precision generally required in NPS monitoring programs, especially those evaluating the effects of treatment or the achievement of TMDL objectives, demand formal laboratory analysis. Laboratories used for NPS monitoring projects may include those operated by state agencies, universities, and private companies.

Specific analytical methods exist for all the water quality variables discussed in this guidance. For all monitoring efforts, analyses should be conducted by accepted laboratory methods. These methods are too numerous to explore in this guidance. There are several resources available to learn about and select appropriate analytical methods, including:

- U.S. EPA Approved Clean Water Act Methods http://www.epa.gov/cwa-methods
- Standard Methods for the Examination of Water and Wastewater, 22nd Edition, American Public Health Association, American Water Works Association, and Water Environment Foundation, http://www.standardmethods.org/ (Rice et al. 2012)
- National Environmental Methods Index (NEMI) https://www.nemi.gov/home/

Select a laboratory to analyze monitoring samples with care. While there is no national certification program for water quality laboratories, most states operate their own certification or registration programs. U.S. EPA operates a Drinking Water Laboratory Certification Program in partnership with EPA regions and states in which laboratories must be certified to analyze drinking water samples for compliance monitoring. Certified laboratories must successfully analyze proficiency testing samples annually, use approved methods, and successfully pass periodic on-site audits. Such certified laboratories may also perform analyses on non-drinking water samples.

When selecting a laboratory, look for one that is certified either by a state program or under the EPA Drinking Water program, one that uses approved methods for analysis, and one that participates in regional comparative proficiency testing programs, if available. In general, it is easier to locate a laboratory to conduct physical and chemical analyses than one to perform analysis of benthic macroinvertebrates, fish, and other aquatic biota. State environmental or natural resource agency biomonitoring programs or university laboratories may be the best bet for bioassessment sample processing. Any laboratory selected, however, should be able to provide documentation of methods and QA/QC protocols used, as well as provide assurance that samples will be handled and processed expeditiously. In making arrangements with the selected laboratory, consider the lab's data approval and reporting system, particularly the likely delays between sample delivery and final data reporting. Long delays in data reporting will inhibit the feedback between land treatment and water quality monitoring that is critically important in watershed project management. Finally, while most water quality laboratories are equipped to analyze water samples for common indicator bacteria like *E. coli*, analysis for pathogens like *E. coli* O157:H7 or *Cryptosporidium* requires considerable expertise generally found only in state health department or private consultant laboratories.

3.7 Land Use and Land Treatment Monitoring

3.7.1 General Considerations

As discussed in section 2.2.1, NPS pollution is generated by activities on the land that vary in location, intensity, and duration. For all monitoring objectives addressed in this guidance (see section 2.1), it will be important to track both land use and land treatment. Note that for the purposes of this guidance, the term "land use" refers not only to the general category of land use or cover (e.g., residential, row crop) but also to land management or source activities (e.g., street sweeping, agrichemical applications, tillage). Similarly, in many cases, the term "land treatment" refers not just to the existence of a specific treatment or BMP (e.g., sediment basin, reduced tillage) but also to the management of the BMP (e.g., sediment basin clean-out, tillage dates, or nutrient application rate, timing, and method). Land use/treatment monitoring encompasses both land use and land treatment.

In general, linking land treatment to water quality response requires both land use/treatment and water quality monitoring. Specific needs for land use/treatment monitoring may differ by monitoring type. For example, assessment monitoring often includes complete spatial coverage of source activities, but temporal variability is not generally addressed because of the short timeframe for problem assessment. Modeling is often used to address the long-term temporal aspects of source activities, including land use changes like conversion of agricultural land to residential use. Evaluating the land uses of a watershed is an important step in understanding watershed condition and source dynamics. Additional details regarding the role of land use in watershed assessment can be found in U.S. EPA's Watershed Planning Handbook (USEPA 2008a).

Understanding of pollutant loading patterns requires information on both the spatial and temporal variability of source activities, particularly when load and wasteload allocations are developed as part of a TMDL. The size of the margin of safety in a TMDL is often directly related to the level of uncertainty associated with the variability of nonpoint source loads (see USEPA 2008a for a discussion of margin of safety [MOS]).

It is necessary to track land use/treatment when planning to attribute water quality trends to activities on the land (see section 2.4.2.4). Because monitoring for trend analysis can continue for decades, costs need

to be factored carefully into decisions about the scope, level of detail, and frequency of land use/treatment monitoring that will be done.

For individual BMP effectiveness monitoring, it is important to document:

- Design specifications of the practice evaluated;
- Degree to which the practice was implemented, maintained, and operated according to specifications;
- Management activities conducted under the scope of the practice; and
- Any situations where the BMP operated under conditions outside of the design range.

For example, it is important to flag any monitoring data collected when the design capacity of a stormwater runoff device is exceeded because performance will often suffer. These same considerations apply to all BMPs to be evaluated at the watershed scale, with the additional proviso that both the spatial distribution and interrelationships of BMPs should be addressed.

Existing guidance provides recommendations for tracking the implementation of agricultural, silvicultural, and urban BMPs (USEPA 1997b, USEPA 1997c, USEPA 2001b). This guidance addresses data sources, methods of data collection, temporal and spatial scales of land use/treatment monitoring, monitoring variables, and sampling frequency.

3.7.2 Basic Methods

3.7.2.1 Direct Observation

Personal observations may be the best way to track land use/treatment for plot and field studies. Studies at this scale are frequently visited for equipment monitoring and sample collection, so a good record of source activities can be obtained. It is recommended that a form be developed and used to ensure that tracking is complete and consistent over time (USDA-NRCS 2003). Examples of such forms are shown in Figure 3-49. Advantages of this method include the ability to schedule visits and the fact that the observer controls the quality of data collected.

Agronomic Data Form

Site name				

MANURE APPLICATION				
Date	Field # (map)	Amt applied (spreader #, loads)	Date incorporated	Comments

Date	Field # (map)	Crop or stocking rate	Activity (till, plant, harvest, etc.)	Comments

Figure 3-49. Examples of agricultural activity data recording forms

Other forms of direct observation include windshield surveys such as those performed by the Conservation Technology Information Center (www.ctic.purdue.edu/CRM/) (CTIC 2016). For some applications, photography can be an important tool. At an edge-of-field monitoring station, an automated digital camera can be installed to take periodic photographs looking up into the drainage area to record crop growth or other visible information. A detailed discussion of the use of photo points for monitoring is presented in chapter 5.

Disadvantages of direct observation methods include the potential for bias due to the observer's lack of understanding of management activities, scheduling that misses important events, and the inability to assess rate or quantity information based only on observation (USDA-NRCS 2003).

3.7.2.2 Log Books

Log books can be given to land owners and managers to record activities relevant to the monitoring study (USDA-NRCS 2003). An advantage of this method is that the same individual who is responsible for the activity does the reporting. However, it is difficult to guarantee compliance or consistent reporting across individuals.

3.7.2.3 Interviews

For interviews, as for log books, reporting is performed by the individual responsible for the activity. When conducted in person, interviews also offer the opportunity to gather additional information of

importance to the study. Disadvantages include the potential for less than complete reporting of information by the person interviewed, as well as potentially inadequate or uneven interview skills by those conducting the interviews (USDA-NRCS 2003).

A combination of the log book and interview approach may work well in small watersheds with a relatively small number of participants. A Vermont project (Meals 2001) successfully used a combination of log books distributed to watershed farmers with an annual interview to collect the logbook and record other information. Interviews were conducted by a local crop consultant who was known and trusted in the region.

3.7.2.4 Agency reporting.

USDA maintains data on conservation practices implemented with USDA cost-share funds or technical assistance. The utility of this information is limited for watershed projects, however, because Section 1619 of the Food, Conservation, and Energy Act of 2008 (section 1619) provides that USDA, or any contractor or cooperator of USDA, may not generally disclose farm-specific information. Exceptions to this prohibition include the disclosure of such information with consent of the producer or owner of the land and statistical or aggregate summaries of the data by which specific farms are not identifiable. Publicly-available data are typically aggregated at the county level and some implementation is not reported due to confidentiality restrictions. In addition, cumulative implementation is difficult to ascertain because maintenance and operation of practices is not tracked. Note also that the information in the system is verified and finalized annually, so data within a current year may be incomplete or inaccurate.

State-level information on USDA conservation programs can be obtained through the RCA Report – Interactive Data Viewer. This information may be useful during the project planning phase to determine the level of program activity and degree to which specific practices are implemented in the state. Farm-specific data, however, would need to be obtained directly from the producer or owner of the land or through a section 1619 agreement with USDA. Hively et al. (2013) describe in detail several section 1619 agreements established within the Chesapeake Bay watershed.

There are also several survey-based inventories of land use information, including USDA's National Resources Inventory (NRI) and the Census of Agriculture (USDA-NASS 2012). Because of confidentiality requirements, the Census of Agriculture does not disclose information on animal populations, crop acreage, or the like for counties with fewer than four individual producers. Data for such non-disclosed counties may need to be estimated, using a variety of approaches (see section 3.7.6).

Other specialized land use datasets include NOAA's Coastal Change Analysis Program's (C-CAP) nationally standardized database of land cover and land change information for the coastal regions of the U.S. Various historical GIS datasets are also available, including the National Land Cover Data and USGS's Land Use and Land Cover data (USEPA 2008a). GIS data for mapping human population are provided by the U.S. Census Bureau through the TIGER (Topologically Integrated Geographic Encoding and Referencing) program. TIGER data consist of man-made features (such as roads and railroads) and political boundaries. Population data from the 2010 Census can be linked to the TIGER data to map population numbers and density for small (census blocks) and large areas (counties and states). In addition, a number of states and counties also have statewide or local land use and land cover information available.

3.7.2.5 Remote Sensing

The basic categories of remote sensing are described in existing guidance (USEPA 2008a). Aerial imagery includes images and data collected from an aircraft and involves placing a sensor or camera on a fixed-wing or rotary aircraft. Space-based imagery includes images and data collected from space-borne satellites that orbit the earth. A wide range of remote sensing datasets are available for free or at low cost, including data products at the USGS's National Map Viewer and Download Platform or Earth Resources Observation and Science (EROS) data center. Other datasets include Landsat data, elevation, greenness, "Nighttime Lights," and coastal and Great Lakes Shorelines (USEPA 2008a). In some regions, FSA conducts annual low-altitude aerial photography to assess compliance with crop insurance programs. If this photography can be accessed with appropriate permissions, it can provide an annual record of crops grown, changes in field boundaries, land development, and other features.

Commercial web-based resources such as Bing Maps and Google Earth can be useful tools for land use monitoring. Although the date of the imagery in these or other resources may not exactly match what is required for a specific project, features such as roads, farmsteads, rivers, and lakes are readily apparent and general land use types (e.g., urban, agriculture, or forest) can be identified and mapped in preparation for acquisition of more current detailed data.

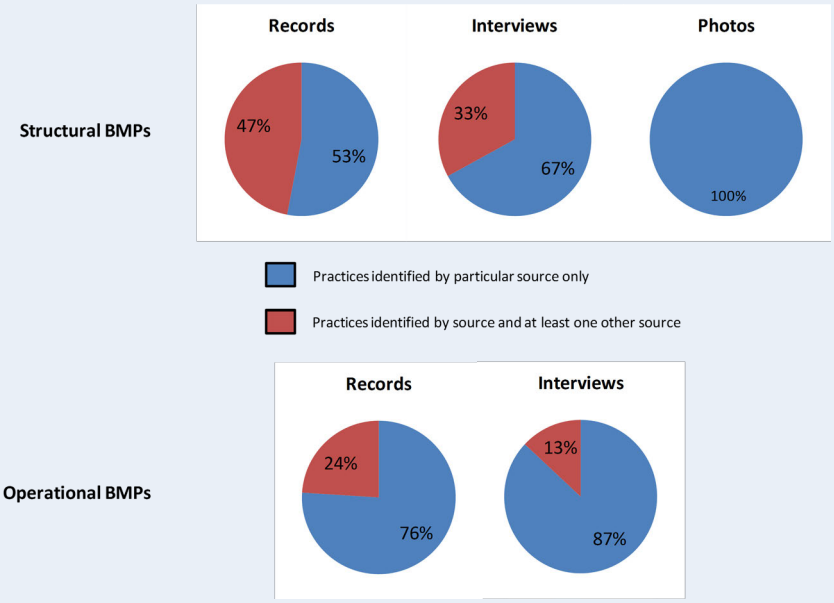
Remote sensing can be useful for tracking practices and land management that are monitored visually. For example, cover crops are easily identified with remote sensing, but whether the cover crops have been fertilized is not easily identifiable. McCarty et al. (2008) used remote sensing technologies to scale point measurements of BMP effectiveness from field to subwatershed and watershed scales, demonstrating that optical satellite (SPOT-5) data and ground-level measurements can be effective for monitoring nutrient uptake by winter cover crops in fields with a wide range of management practices. Hively et al. (2009a and 2009b) combined cost-share program enrollment data with satellite imagery and on-farm sampling to evaluate cover crop N uptake on 136 fields within the Choptank River watershed in Maryland. Annual cost-share program enrollment records were used to locate cover crop fields and provide agronomic management information for each field. Satellite imagery from December and March was used to measure pre-winter and spring cover crop biomass, respectively. Data collected simultaneously from fields were used to convert satellite reflectance measurements to estimates of biomass and nutrient uptake, thus providing a means to estimate aboveground biomass and N uptake estimates for all fields enrolled in the cover crop program.

Locating Best Management Practices by Three Methods
Eagle Creek Watershed, IN — NIFA-CEAP Watershed Project

Objective: **To assess the effects of BMPs on water quality, researchers needed to identify all BMPs implemented in an agricultural watershed since 1995 under a variety of state and federal programs**

Three approaches based on different data sources were used:

- Examination of government records (NRCS, FSA, Indiana Dept. of Env. Mgt.)
- Interviews with producers (structural and operational, polygon and line format)
- Analysis of aerial photography (structural only)



Observations:

1. NRCS data required processing to eliminate double-counting because each point potentially represented multiple practices. After eliminating all the double-counting, 107 structural practices were reduced to 48 standard practices and 299 operational BMPs to 84 distinct practices.
2. Remote sensing picked up only 27 structural practices and no operational practices
3. Producer interviews detected 47 structural practices and 185 operational practices
4. Using all three sources of information, 94 structural practices and 215 operational practices were identified.
5. 53% of the structural practices were identified by government records, while 67% were identified through producer interviews.
6. Operational practices were identified in government records 76% of the time relative to 87% from producer surveys.
7. Researchers found that:
   - Government records identified the majority of BMPs, but were incomplete and difficult to obtain
   - Interviews were information-rich but time-consuming to conduct
   - Photos were effective to confirm and supplement records and interviews
   - Combined data collection techniques provided a clearer picture of conservation practices in the watershed compared to any single approach.

(Grady et al. 2013)

3.7.3 Temporal and Spatial Scale

Land use/treatment monitoring should address the entire area contributing to flow at the water quality sampling point. Depending on the specific study area and monitoring design, some parts of a larger area may be emphasized more than other parts. For example, land nearest to the sampling point can sometimes have a major effect on the measured water quality, so these areas must be monitored carefully. Thus, the spatial coverage of land use monitoring may range from a single field (or portion of a field) up to an entire river basin.

In designing a land use/treatment monitoring system, it is logical to begin with the assumption that the temporal scale of land use/treatment monitoring should match that of the water quality monitoring when the data are to be combined for analyses. Data from weekly composite water quality samples, for example, would be associated with weekly measures of source activity. However, this design should be tempered by understanding the inherent variability of what is being measured (see section 3.7.5). Some metrics of land use and land treatment do not in fact vary on a weekly time scale. It would be wasted effort, for example, to determine and record the crop present in an agricultural field each week during a single growing season or note that a residential subdivision is composed of moderate density detached homes. On the other hand, some highly transient land management activities are very critical to water quality. Manure application on cropland, tillage operations, and street sweeping are examples, and weekly records of such phenomena would be important. Still other land management activities may be important to identify exactly in time and magnitude, for example in relation to a storm event. Herbicide losses from cropland, for example, are strongly influenced by proximity of application to the first few runoff events; pollution potential of pasture runoff may be influenced by the number of grazing animals around the time of major runoff events.

A multi-level land use/treatment monitoring approach can address these multiple temporal concerns:

- **Characterization:** an initial snapshot of land use/land cover, focusing on relatively static parameters (at least relative to the project period) such as water bodies, highways, impervious cover, and broad patterns of urban, agricultural, and forest land uses;
- **Annual:** an annual survey for annually-varying features such as crop type;
- **Weekly:** weekly observations or log entries to identify specific dates/times of critical activities like manure or herbicide applications, tillage, construction, and street sweeping; and
- **Quantitative:** data collection on rates and quantities (e.g., nutrient or herbicide application rates, number of animals on pasture, logging truck traffic).

The guiding principle of timing is to collect land use/treatment data at a fine enough time resolution to be able to (at least potentially) explain water quality observations (e.g., a spike in P concentration) as they occur.

It is important to note that associations between land use/treatment observations and water quality patterns can be confounded by the timing of the source activities (USDA-NRCS 2003). For example, road salt is applied under icing conditions, while wash off tends to occur during periods of thawing or rainfall. Matching weekly water quality and land use/treatment in this case could result in associating high salinity levels with periods of no road salt application. As another example, nutrient concentrations peak during wet periods, but manure is not usually applied when fields are muddy. Using weekly data, high nutrient concentrations would be associated with periods of no manure application. An understanding of pollutant pathways and lag time (section 6.2) and some creative data exploration are often needed to effectively

pair land use/treatment observations with water quality data, but this becomes more difficult moving from the BMP level to the watershed scale. Such issues may be addressed by pairing annual water quality data with annual land use/treatment data (Meals 1992); although fine-scale relationships may be lost by this data aggregation, broad patterns of the influence of land use on water quality may be established.

3.7.4 Monitoring Variables

The appropriate set of land use/treatment variables for any monitoring plan will depend on the monitoring objectives, monitoring design and characteristics of the watershed or site to be monitored. The set of land use/treatment variables needed for problem assessment is usually broad (USEPA 2008a), whereas the set of variables for BMP effectiveness monitoring is tailored to the BMP and the conditions under which it is being evaluated.

Table 2-2 in section 2.2.2 illustrates an important first step in selecting land use/treatment variables appropriate for the monitoring plan. The next step involves selecting the specific water quality variables and matching those with specific land use/treatment variables for which a relationship is likely. Table 3-13 shows examples of pairing water quality and land use/treatment variables.

Table 3-13. Relationship of water quality and land use/land treatment variables

Source	Water Quality Monitoring Variable	~Weekly Land Use/Treatment Monitoring Variables	~Annual Land Use/Treatment Monitoring Variables
Cropland Erosion	Suspended Sediment	• Date of tillage operations; • Tillage equipment used; • Crop canopy development; • Cover crop density	• Acreage (and percentage) of land under reduced tillage; • Acreage (and percentage) served by terrace systems; • Acreage (and percentage) of land converted to permanent cover; • Linear feet (and percentage of linear feet) of watercourse protected with riparian buffers
Agricultural Cropland	Total Nitrogen	• Manure and/or fertilizer application rates; • Manure and/or fertilizer forms; • Date of manure and/or fertilizer application; • Manure and/or fertilizer application methods	• Number (and percentage) and acreage (and percentage) of farms implementing comprehensive nutrient management plans; • Annual fertilizer and manure N applications per acre; • Legume acreage; • N fertilizer sales
Urban	Stream Flow	• Operation and maintenance of stormwater system; • Functioning of stormwater diversions or treatment devices	• Percent impervious cover; • Acreage (and percentage) served by water detention/retention; • Number and area of rain gardens or other infiltration practices

"~Weekly" variables are those that must be monitored frequently to record the exact date or quantity associated with the metric. "~Annual" variables can be determined less frequently as they generally remain constant within a crop year.

3.7.5 Sampling Frequency

As discussed briefly in section 3.7.3, land use/treatment data can be either static or dynamic (USDA-NRCS 2003). Static land use/treatment data such as soil type and slope do not generally change with time,

but dynamic land use/treatment data can vary with time. Examples of dynamic land use/treatment data include the number of animals, crop rotations, cover crops, undisturbed area, nutrient and pesticide applications, road salting, and irrigation schedules.

Sampling frequency will vary depending on the study design and source activity. For BMP effectiveness studies at the plot or field scale, observations should be made each time the site is visited (USDA-NRCS 2003). It is possible to easily observe the entire study area at these scales, but observations made at monitoring stations for larger-scale projects, although important to do, will not cover the entire study area. The frequency for sampling dynamic data will vary depending on the type and magnitude of the variable's impact on measured water quality. For example, construction activities occur on a daily basis at any given construction site, but there are construction phases that are more important than others (e.g., site clearing) and therefore warrant closer attention. The availability of records should also be considered when determining sampling frequency. Producers under many nutrient management plans, for example, must keep field-by-field records of manure and chemical nutrient applications, so sampling can theoretically be done on an annual basis assuming that the records are clear and accurate.

3.7.6 Challenges

There are many challenges associated with tracking land use/treatment, including gaining access to locations for direct observation or communication with landowners or managers. Obtaining cooperation on field logs also represents a major hurdle in many cases, especially when confidential business information is involved. At the watershed scale, the task of checking all source activities of potential interest can be difficult logistically, labor intensive, and complicated in a mixed use watershed where different areas of expertise may be needed to track a wide range of source activities.

Data confidentiality can present major challenges to monitoring land management in a watershed project. Confidentiality applies at many levels, from individual landowners participating in USDA cost-share programs through their local NRCS district to county or watershed-level data reported in the Census of Agriculture. In small projects, a good way to overcome this obstacle is to obtain permission from the landowner; with such permission, NRCS and FSA records of BMP implementation will be accessible. In some field-scale projects, it may be possible to have the cooperating landowner(s) sign a release at the beginning of the project to allow access to their records, including nutrient management plans, participation in cost-share programs, BMP installation, etc.

Dealing with larger scale agency data is more problematic. As noted previously (section 3.7.2.4), data reported by the Census of Agriculture are not disclosed if a limited number of producers are present in a county or watershed. This data gap presents a challenge to determining basic characteristics of a county or watershed such as cropland acres or animal populations. There are, however, some helpful approaches to estimate the undisclosed data. For example, if dairy cow numbers are not disclosed for a county of interest, it is possible to add up the numbers for reported counties, subtract that sum from the state total to arrive at a number for the "remainder" dairy cows. If data from more than one county are non-disclosed, the "remainder" animals can be apportioned by county area, cropland acres, or other reported variable. Although such procedures are cumbersome and add uncertainty, they often represent the best or only source of data for a project area.

Estimation of non-disclosed Census of Agriculture data Nutrient Use Geographic Information System (NuGIS) International Plant Nutrition Institute

Objective: **A major national study of fertilizer nutrient balance by county needed to derive estimates of fertilizer nutrients applied and removed in harvested crops for each U.S. county**

Standard procedures for estimating data missing due to non-disclosure in Census of Agriculture were developed:

- *When Census of Ag Production data for a commodity were not disclosed for some counties in a state, subtracting the sum of disclosed production for a commodity from the state total production for that same commodity yielded a remainder – the 'State Production Remainder' – that represents the sum of production in non-disclosed counties for that commodity. We apportion the State Production Remainder for this commodity to each county in a state with non-disclosed production for this commodity, based on each county's harvested acres of this commodity as reported in the Census of Ag or as estimated.*
- *For each commodity, the amount of State Production Remainder that is apportioned to each county with a non-disclosed production value was calculated using a 'Production to Harvested Acres coefficient'; this could also be thought of as an estimated yield. This coefficient was calculated, for each commodity, in each state, by dividing the (State Production Remainder) by the (Sum of Harvested Acres in counties with non-disclosed Production). The county crop production was then calculated using:*

  (County Total Cropland Acres) X (Harvested Acres to Total Cropland Acres coefficient).

Example:

State Production Remainder for Corn =
(State total production of corn) - (sum of corn production in counties with data disclosed)

2 million bu corn – 1 million bu corn = 1 million bu corn

State Production to Harvested Acres coefficient for corn =
(State Production Remainder for Corn) /
(Sum of Harvested Acres of Corn in counties with non-disclosed production of corn) =

(1 million bu) / (5,000 Harvested Acres of Corn) = 200 bu corn / harvested acre of corn

Estimated Production of Corn for County A =
(Harvested Acres of Corn in County A) X (Production to Harvested Acres Coefficient for corn) =
(3,000 Harvested acres) X (200 bu corn / harvested acre) = 600,000 bu of corn production

(IPNI 2010)

3.8 Special considerations for pollutant load estimation

Because of the central role pollutant loads and load reduction targets play in many watershed projects, especially those with TMDLs, the accuracy of load estimates is very important to all project stakeholders. Further, the potentially high relative cost of monitoring for load estimation (see chapter 9) places a premium on cost-effectiveness. This section combines many of the observations made in this chapter about monitoring for load estimation in one place to provide basic guidelines and considerations for this special type of monitoring. Richards (1998) provides a comprehensive discussion of pollutant load estimation techniques and is the source of much of the information presented here.

Pollutant flux (see Box) varies tremendously with both flow and pollutant concentration. Because we cannot measure flux directly or continuously, we usually compute unit loads (e.g., daily or monthly) as the product of discharge and pollutant concentration, then sum these unit loads to produce an estimate of annual load.

Basic Pollutant Load Terms

Flux – instantaneous loading rate (e.g., kg/sec)

Flow rate – instantaneous rate of water passage (e.g., L/sec)

Discharge – quantity of water passing a specified point (e.g., m3)

Load – mass of substance passing a specified point (e.g., metric tons).

The following steps are recommended to plan a monitoring effort for load estimation:

1. Determine whether the project goals require knowledge of load, or if goals can be met using concentration data alone. In many cases, especially when trend detection is the goal, concentration data may be easier to work with and be more accurate than crudely estimated load data. However, some concurrent hydrologic/meteorologic data (flow, stage height, rainfall, etc.) are often needed for some aspect of any watershed study.

2. If load estimates are required, determine the accuracy and precision needed based on the uses to which they will be put. This is especially critical when the purpose of monitoring is to look for a change in load. It is foolish to attempt to document a 25 percent load reduction from a watershed program with a monitoring design that gives load estimates ± 50 percent of the true load (see Spooner et al. 2011).

3. Decide which approach will be used to calculate the loads based on known or expected attributes of the data. This decision will also lead to choices on monitoring equipment (e.g., whether an automatic sampler will be used). See section 7.9.2 for a discussion of approaches to load estimation and see below for a discussion of sampling equipment.

4. Use the precision goals from Step 2 to calculate the sampling requirements for the monitoring program. Sampling requirements include both the total number of samples and the distribution of the samples with respect to some auxiliary variable such as flow or season. See section 3.4 and below for information on sampling frequency and distribution.

5. Calculate the loads based on the samples obtained after the first full year of monitoring, and compare the precision estimates (of both flow measurement and the sampling program) with the initial goals of the program. Adjust the sampling program if the estimated precision deviates substantially from the goals. See *Interval Estimation* (p. 4-18 of the 1997 guidance [USEPA 1997a]) or Spooner et al. (2011) and section 3.4.2 for information relevant to this step.

3.8.1 Sample Type and Sampling Equipment

The basic approaches for load estimation described in section 7.9 of this guidance are numeric integration, regression, and ratio methods. With numeric integration, the goal is to collect representative concentration samples for each sampling interval which is typically defined either by the calendar (e.g., daily, weekly) or by the volume of flow that passes by the sampling point. In other words, there are no data gaps. For both the regression and ratio methods, it is assumed that a strong relationship exists between concentration and flow and that there will be sampling intervals for which only flow is measured (i.e., no concentration samples taken). With the regression approach, the missing concentration values are then estimated from the relationship of flow and concentration (when concentration samples were taken). The ratio approach assumes that flow is measured for each sampling interval and that daily loads are calculated for those days when concentration samples are taken. A flow ratio (annual flow/flow for days with concentration samples) is then used in combination with a bias correction factor (to account for correlation between discharge and load) to estimate annual pollutant load. Both the regression and ratio methods can be performed using annual or seasonal relationships. These relationships may change over time, particularly in cases where BMPs are implemented, so it is important that the relationships are re-examined at least annually.

Autosamplers are typically required for numeric or composite integration because of the large number of concentration samples needed. Grab sampling is typical for both the regression and ratio methods, but autosamplers can be used. Continuous or near-continuous flow measurement is required for all three methods but in some cases flow data are obtained from others (e.g., USGS).

Section 3.2 describes many options for sample type, the simplest of which is a grab sample. The specific type of sample appropriate for each project will depend on the details of the load estimation objective. For example, it may be desirable to track the variability of both concentration and load during the sampling interval. In this case, multiple discrete samples over time would be preferred over composite samples; the cost for sample analysis, however, would increase considerably. Where fluctuations within the sampling interval are not of interest, composite samples would be recommended. Flow-proportional samples are recommended for load estimation in these cases.

3.8.2 Sampling Frequency and Timing

Sample type is an essential consideration involved in sampling for good load estimation but sampling frequency and sample distribution over time are equally important. The selection of sampling frequency required for accurate estimation of pollutant loads is more challenging than for concentration because load is a product of concentration and flow, both of which usually vary significantly. Furthermore, in NPS situations, because the majority of the annual pollutant load often occurs in a few major events, the choice of *when* to sample is also critical.

Ideally, the most accurate approach to estimating pollutant load would be to sample very frequently and capture all the variability. Flow is relatively straightforward to measure continuously (see Meals and Dressing 2008 and section 3.1.3.1), but concentration is expensive to measure and in most cases impossible to measure continuously. It is therefore critically important to choose a sampling interval that will yield a suitable characterization of concentration. Strategies for determining sampling frequency and timing for accurate load estimation are described below; see Richards (1998) for additional information.

Sampling frequency determines the number of unit load estimates that can be computed and summed for an estimate of total load. Using more unit loads increases the probability of capturing variability across the year and not missing an important event; in general, the accuracy and precision of a load estimate

increase as sampling frequency increases. For example, the top panel in Figure 3-50 shows load estimated from weekly sampling superimposed on idealized daily load data. The bottom panel shows results plotted from monthly and quarterly sampling on top of the same daily load data. The weekly data appear to capture much of the variation of the daily series, but the monthly series does much more poorly. Quarterly sampling clearly misses many important peaks and overstates periods of low flux.

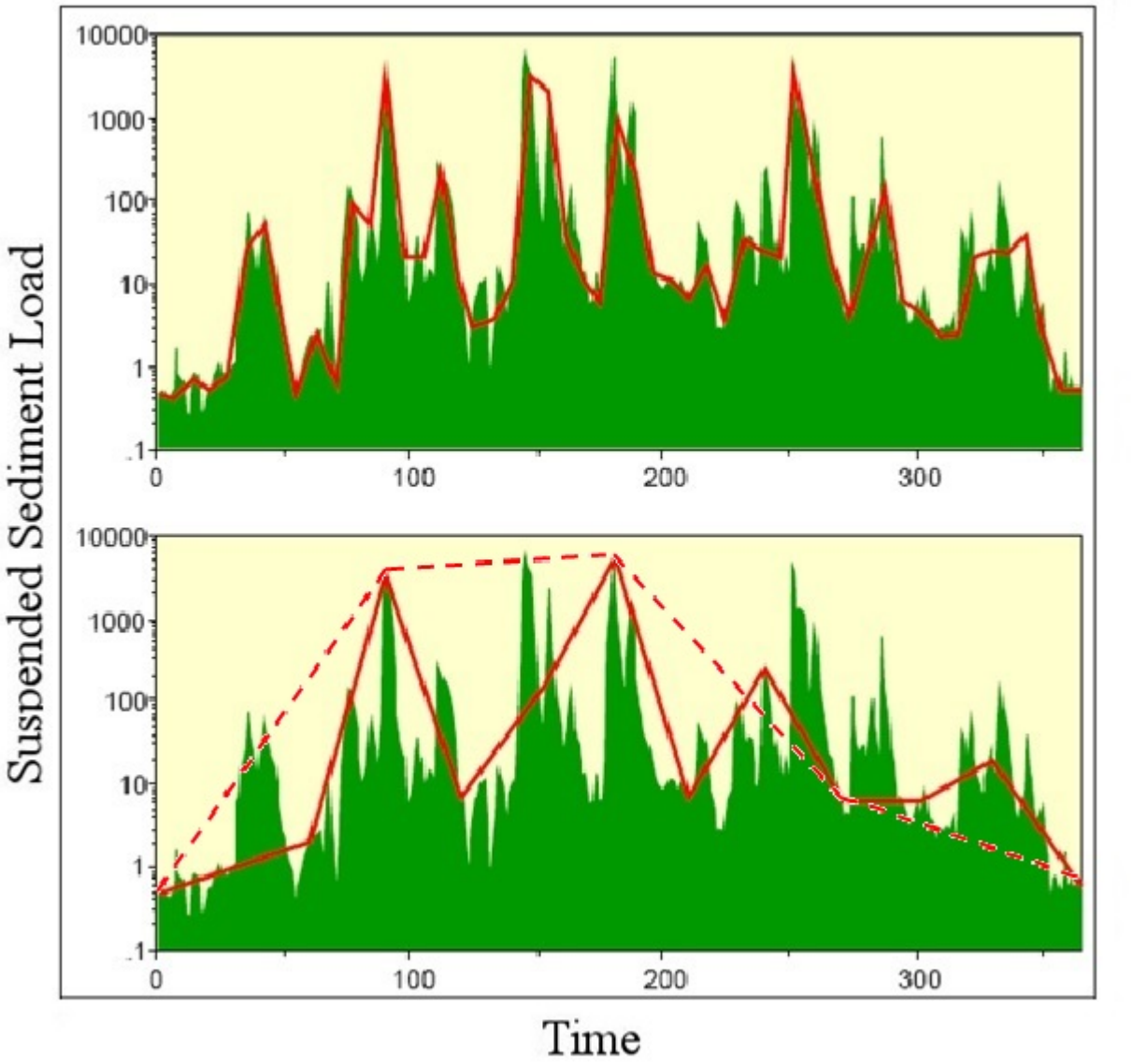


Figure 3-50. Weekly (top panel) and monthly and quarterly (bottom panel, solid and dashed lines, respectively) load time series superimposed on idealized daily load time series (adapted from Richards 1998)

There is a practical limit to the benefits of increasing sampling frequency, however, due to the fact that water quality data tend to be autocorrelated (see section 7.3.6). The concentration or flux at a certain point today is related to the concentration or flux at the same point yesterday and, perhaps to a lesser extent, to the concentration or flux at that spot last week. Because of this autocorrelation, beyond some point, increasing sampling frequency will accomplish little in the way of generating new information. This is

usually not a problem for monitoring programs but can be a concern when electronic sensors are used to collect data nearly continuously.

The choice of *when* to collect concentration samples is critical. Most NPS water quality data have a strong seasonal component as well as a strong association with other variable factors such as precipitation, streamflow, or watershed management activities such as tillage or fertilizer application. Selecting when to collect samples for concentration determination is essentially equivalent to selecting when the unit loads that go into an annual load estimate are determined. That choice must consider the fundamental characteristics of the system being monitored. In northern climates, spring snowmelt is often the dominant export event of the year; sampling during that period may need to be more intensive than during midsummer in order to capture the most important peak flows and concentrations. In southern regions, intensive summer storms often generate the majority of annual pollutant load; intensive summer monitoring may be required to obtain good load estimates. For many agricultural pesticides, sampling may need to be focused on the brief period immediately after application when most losses tend to occur. In arid areas, it may be more appropriate to collect storm composites, focusing sampling efforts on the normal wet periods. Regardless of the approach chosen, it is essential that loads are calculated after the first year in accordance with Step 5 above to determine if precision needs are met.

For both the regression and ratio approaches, determination of sampling frequency may assume a normal distribution for concentration and random sampling. Several formulas are available to calculate the number of samples (random or within strata) required to obtain a load estimate of acceptable accuracy based on known variance of the system (see chapter 2 of the 1997 guidance). Stratification may improve the precision and accuracy of the load estimate by allocating more of the sampling effort to the aspects which are of greatest interest or which are most difficult to characterize because of great variability such as high flow seasons.

3.8.3 Planning and Cost Considerations

As described here, the sampling regime needed for load estimation must be established in the initial monitoring design, based on quantitative statements of the precision required for the load estimate. In many cases, the decision to calculate loads is sometimes made after the data are collected, often using data collected for other purposes. At that point, little can be done to compensate for a data set that contains too few observations of concentration, discharge, or both, collected using an inappropriate sampling design. Many programs choose monthly or quarterly sampling with no better rationale than convenience and tradition. A simulation study for some Great Lakes tributaries revealed that data from a monthly sampling program, combined with a simple load estimation procedure, gave load estimates which were biased low by 35 percent or more half of the time (Richards and Holloway 1987).

Monitoring programs often struggle with a conflict between the number of observations a program can afford and the number needed to obtain an accurate and reliable load estimate. Most use flow as a means to estimate the best intervals between concentration observations. For example, planning to collect samples every x thousand ft^3 of discharge would automatically emphasize high flux conditions while economizing on sampling during baseflow conditions.

It is possible, however, that funding or other limitations may prevent a monitoring program from collecting the data required for acceptable load estimation. In such a case, the question must be asked: is a biased, highly uncertain load estimate preferable to no load estimate at all? Sometimes the correct answer will be no.

3.9 Data Management

3.9.1 General considerations

Data management can be defined as the development, execution and supervision of plans, policies, programs and practices that control, protect, deliver, and enhance the value of data and information assets (Mosley et al. 2009). Small, short-term monitoring projects can often set up and operate their own effective data management system using basic tools like spreadsheets and paper files. Depending on the magnitude and duration of the monitoring project, it may be advisable to go beyond immediate local data storage and reporting practices and participate in and comply with ongoing USEPA data management programs (e.g., USEPA 2010). Regardless of the magnitude of the monitoring effort, data management must be part of initial project planning.

Data management planning should be an integral part of developing a monitoring plan as reflected by its inclusion as a Group B element in QAPPs (USEPA 2001a). The aspects of data management to be described in a QAPP include the path of the data from their generation to their final use or storage, the standard record-keeping procedures, document control system, and the approach used for data storage and retrieval on electronic media. In addition, the control mechanism for detecting and correcting errors and for preventing loss of data during data reduction, data reporting, and data entry to forms, reports, and databases are to be described in the QAPP. Examples of any forms or checklists to be used are also required, as are descriptions of all data handling equipment and procedures to process, compile, and analyze the data. This includes procedures for addressing data generated as part of the project as well as secondary data from other sources. Required computer hardware and software and any specific performance requirements for the hardware/software configuration used are to be described. Data analysis software options are described in chapter 7.

3.9.2 Data acquisition

Sections 2.1 through 3.7 and chapters 4-5 address experimental design, sample collection, and sample analysis methods for a wide range of nonpoint source monitoring projects. The data generated by these monitoring projects must be collected (data acquisition) and transferred to the data management system for storage and analysis.

Field and laboratory procedures may include the use of field books or data entry sheets to record observations and measurements and either paper or electronic data report forms. The transcription of data reported in these fashions into a database is a potential source of typographic errors, switched digits, and other errors in data entry. It is crucial that all data be error-checked after entry into electronic forms, but before analysis and reporting. Finding errors in a dataset after analysis and reporting is underway can be very frustrating.

Newer methods of data acquisition include the use of data loggers (either external loggers that record multiple data streams or loggers directly built into sensor devices), laptops, tablets, and smartphones to allow direct acquisition, transmission, and entry of data to electronic media. An advantage of using data loggers is that manual data entry and the associated transcription errors are avoided (USDA-NRCS 2003). Remote access allows direct transfer of field data from a data logger to the main data storage site. One disadvantage of data loggers is that their storage capacity is limited; once full, new data may not be recorded or older data may be overwritten and thus lost. It is strongly recommended that monitoring protocols include prompt and routine downloads of data from field data loggers.

Not all data are generated directly by the project. Element B-9 of a QAPP addresses data obtained from non-measurement sources such as computer databases, programs, literature files, and historical data bases (USEPA 2001a). Whenever data are obtained from other sources, it is important to determine the sufficiency of the data for project purposes (USEPA 2008a). One of the challenges of using GIS data, for example, is the need to ground-truth and fill gaps in the data layers (USDA-NRCS 2003). Johnson and Zelt (2005) present a method for filling in a data gap of spatial scale in woodland LULC (land-use/land-cover) between the land-cover data available from the 30-m 1990s National Land Cover Dataset (NLCD) and the reach-level data available from the prescribed National Water Quality Assessment (NAWQA) habitat assessment.

Data provided by others may have been collected at different locations, by different methods, or to serve different objectives from those of the current project, so it is important to carefully review the data and methods used for its collection. This situation is a common occurrence in the watershed project planning phase during which projects often must use whatever data are available to characterize problems and suggest actions to solve those problems. The QAPP should include acceptance criteria for the use of such data in the project, as well as any data use limitations (USEPA 2001a).

3.9.3 Data storage

Data storage includes both manual and computerized technologies (USDA-NRCS 2003). All field and laboratory notebooks must be fully documented and stored safely, and all data contained in the notebooks should be backed up in paper or digital form, perhaps as scanned images.

A data inventory is important for monitoring projects, particularly those focused on problem assessment. Information on ways to organize and manage a data inventory is provided in existing guidance (USEPA 2008a). Naming and labeling conventions should be established, and metadata (e.g., where, how, why, when and what was monitored) should be included with all datasets.

Spreadsheets might be adequate for data generated by small projects, but a relational database is usually preferable for more complex projects involving many sites or variables (USEPA 2008a). A relational database houses data, metadata (information about the data), and other ancillary information in a series of relational tables including station information, sample information, analyses, methods used, and quality control information.

All computerized data and electronic project files should be backed up using one of many options, including USB flash drives, external hard drives, CDs, remote servers via File Transfer Protocol (FTP), and commercial data storage systems available on-line. All media have their advantages and disadvantages. As technology changes, computerized data should be copied to the latest media using the latest software. For archival purposes, data storage as paper printouts may be a preferred choice; consider that 1985 data archived on 5.25-in floppy disks would be next to useless today. Daily backup of computerized data and electronic project files is recommended. Where practical, backups should be stored offsite for protection against theft, fire or water damage. Today, with the proliferation of relatively inexpensive and free options for data backups, there is little excuse for losing data due to computer failure once the data has returned from the field or laboratory.

3.10 Data Reporting and Presentation

3.10.1 General considerations

Data reporting and presentation occur at multiple levels in many forms to address a wide range of audiences and purposes. Communication with stakeholders is often best done on a frequent, informal basis, whereas communication with outside audiences is more commonly accomplished via presentations at professional meetings or publications of project findings.

Funding agencies generally include reporting requirements in their grants or contracts. Some include requirements to upload the data to repositories such as EPA's STORET (http://www.epa.gov/storet/). States receiving section 319 grants are required to use GRTS (Grants Reporting and Tracking System) to report specific nationally mandated data elements (USEPA 2013).

3.10.2 Communicating with Stakeholders

Project managers should schedule regular meetings with stakeholders to present available data and discuss both successes and failures. Project staff will often find that stakeholders have information, ideas, and resources they need to improve the project or make their objectives easier to accomplish. Quarterly meetings are recommended, so those collecting and analyzing the data should examine the data frequently to be familiar with the current status of the project and to identify and fix problems. The USGS, for example, recommends that field and laboratory results be examined as soon as possible, preferably before the next sample-collection field trip (Wilde 2005). Results indicating potential bias in the data may trigger needed changes in equipment, equipment-cleaning procedures, or field methods used.

Communicating with groups of individuals with varied levels of understanding and different learning styles requires a multimedia approach that includes written materials, audio-visual presentations, and face-to-face communication. Simple quarterly reports with easily interpreted graphs, summary tables, and maps will enhance the communication. Reports should highlight observed patterns and both raw data and metadata should be attached for those in the audience with more advanced understanding of project data. A particularly powerful tool for presenting information to any audience is a Geographic Information System (GIS) that can be used to create watershed maps and display a variety of spatial information (USEPA 2008a). Users can display selected data and a combination of spatial coverages tailored to the specific audience and venue.

3.10.3 Final reports

Final reports are an essential element of all monitoring projects, but experiences of the Rural Clean Water Program (USEPA 1993a) and similar watershed programs show that project budgets frequently do not provide sufficient resources for final data analysis and reporting. One way to address this problem is to require quarterly reports and meetings as described in section 3.10.2. A major hurdle associated with final reports is the task of pulling together all project data and performing the final analyses. This burden is reduced substantially if reports and analyses have been generated on a regular basis since the beginning of the project.

The basic elements of a project report are the title, abstract, introduction, body, summary and conclusions, references, forward, preface, appendices, glossary, tables, and illustrations (USGS 2008). The introduction should include the purpose and scope of the report, and will usually include background information pertinent to the study. The body of the report includes the purpose of the study, data summaries, and the analyses and interpretation of the data. The summary and conclusions pull together

the major results and conclusions described in the body. A concise Executive Summary is useful as a pull-out section to distribute project results to a wide audience.

State and federal agencies have their own guidelines and reporting requirements. Professional publications and journals specify reporting requirements at their websites.

3.11 References

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